Written Submission

for the
Royal Society for the Protection of Birds
and Suffolk Wildlife Trust

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Planning Act 2008 (as amended)

In the matter of:

Application by NNB Generation Company (SZC) Limited for an Order Granting Development Consent for

The Sizewell C Project

Planning Inspectorate Ref: EN010012
RSPB Registration Identification Ref: 20026628
Suffolk Wildlife Trust Registration Identification Ref: 20026359

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1. INTRODUCTION

The RSPB

1.1 The Royal Society for the Protection of Birds (the RSPB) was set up in 1889. It is a registered charity incorporated by Royal Charter and is Europe's largest wildlife conservation organisation, with a membership of more than 1.1 million. The RSPB manages 220 nature reserves in the UK covering an area of over 158,725 hectares. The Society attaches great importance to the conservation of the 'Natura 2000' network (made up of Special Protection Areas (SPAs) and Special Areas of Conservation (SACs)), and the national network of Sites of Special Scientific Interest (SSSIs) notified by Natural England.

Suffolk Wildlife Trust

1.2 Suffolk Wildlife Trust has led local nature conservation efforts in Suffolk for five decades and has saved some of the county's most important wildlife sites as nature reserves. In the past eight years the Trust has spent over £7 million buying land for wildlife and currently cares for nearly 8000 acres of some of Suffolk's most inspiring wild places.

2. SUMMARY OF OUR CONCERNS

2.1 The Suffolk Coast is a network of coastal, estuarine and freshwater wetlands with shingle beaches, areas of Sandlings heathland, many of these areas designated for their conservation value. The area around Sizewell is particularly valuable, being next to Sizewell Marshes, a nationally protected site with irreplaceable habitats and species, and adjacent to Minsmere, the RSPB's flagship reserve and part of a wider site designated at the highest levels for its conservation value. The proposals for the development of Sizewell C pose significant risks to both these sites, and nature conservation more widely on the Suffolk Coast. Here we summarise the overall potential impacts.

2.2 Minsmere has a unique place in the history of nature conservation in the UK, being the place where the avocet, an iconic species previously lost from the UK, returned to breed and from where it began its recolonisation of the UK's wetlands. Likewise, the marsh harrier, now a familiar sight drifting over reedbeds on the Suffolk Coast and beyond, was reduced to a single pair but made its comeback from here. As well as its bird life, the reserve is important for a range of other species of plants, invertebrates, bats, amphibians and reptiles and has a known (but constantly growing) list of well over 6000 species. This is due to the range of habitats Minsmere contains, including reedbed, open water and scrape, wet grassland, ditches, coastal shingle, woodland, heather heathland and acid grassland.

2.3 Sizewell Marshes is a nationally designated nature conservation site, important for its lowland wet meadows, fens, reedbeds and wet woodlands. It holds populations of breeding wildfowl, rare invertebrates and plants. Important populations of bats also breed, hibernate and forage here, including nationally significant population of rare barbastelle.

2.4 The proposals for Sizewell C incorporate part of Sizewell Marshes, meaning that over 6ha of this SSSI is proposed to be lost entirely and a further 3ha damaged. The loss of nearly 10% of a nationally protected site is of serious concern to us, especially as some of the habitats lost are very hard, if not impossible, to recreate. The proposals for Sizewell C do include habitat compensation sites to try to recreate the habitats lost, but whilst reedbed has been created at Aldhurst Farm, the likelihood of success in recreating fen meadow is still not known and we still do not believe the proposals for fen meadow and wet woodland are adequate to ensure the re-establishment of the specialist plants and invertebrates lost from these habitats.

2.5 The Sizewell C proposals have the potential to significantly affect the habitats and species of Minsmere and Sizewell Marshes, mainly because of the proximity of the planned development to these sites. The construction area would be directly adjacent to the Minsmere South Levels and to Sizewell Marshes, containing important areas of wet grassland and pools for breeding and wintering wildfowl and for foraging marsh harriers. The significant levels of noise and visual disturbance from the construction area and the length of time over which these impacts would be expected (10-12 years) means that any displacement of these species could have a significant effect on their populations. For marsh harriers, from Minsmere foraging on the South Levels to further afield on Sizewell Marshes and the surrounding farmland, this means that they will effectively lose significant parts of their regular hunting grounds, which could affect their ability to provide food for their chicks. Over a period of years, this could affect Minsmere's marsh harrier population numbers. Waders and wildfowl on the South Levels and Sizewell Marshes may also be displaced by construction disturbance, and with significant proportions of Minsmere's populations of these species using these areas, this again could affect population levels in the medium term. We are also concerned that for some elements of the construction, timing is key to avoid impacts – if the timing of the construction of one of the water storage areas in the northern part of the construction area slips into the spring, the noise disturbance from this could affect breeding birds in Minsmere's reedbeds, home to specialists such as the bittern, marsh harrier and bearded tit.

2.6 Bats, including the rare barbastelle, would also be disturbed and displaced by construction – by the presence of lighting, noise and human activity, and the loss of some of the important connecting and feeding habitats between Minsmere and Sizewell as a result of the construction area. Roosting and hibernation sites for barbastelle may also be significantly impacted and it is likely that many unrecorded roosting sites will be lost.

2.7 The wetland areas of Minsmere and Sizewell Marshes are also at risk from changes to their hydrology. The movement of ground and surface water is crucial to maintaining the function of specialist habitats and the species they support. This is why we are so concerned about any potential impacts on Minsmere Sluice – a key structure which controls the drainage of Minsmere and much of the surrounding area. The combination of waterways entering the Sluice means that impacts on one of these waterways has the potential to have much wider effects. During the construction of Sizewell C, the discharge of water to one of these waterways, Leiston Drain, is expected to increase. The amount of water entering the Sluice from this source may mean that it is more difficult for water from the other waterways to drain at the same rate as previously. This could mean that it is not possible to drain Minsmere quickly when spring flooding occurs, meaning that nests of species such as lapwing, redshank, avocet and bittern are flooded out and habitats take longer to recover.

2.8 The Sluice also allows some entry of seawater into Minsmere to help maintain some of the specialist brackish habitats in the ditches and on the scrape. This means that these habitats are vulnerable to any contaminants entering by this route. Sizewell C would discharge hydrazine, a toxic chemical used as an anti-foulant in the cooling water system, during the later stages of construction and throughout its operation. At certain states of the tide it is possible that low levels of hydrazine would enter Minsmere via the Sluice. As yet, we do not know what the long-term impacts of this would be on the habitats and species of the scrape and ditches.

2.9 As the construction site for the power plant currently consists of soft, wet soils, the site will need to be drained and a hydrological cut-off wall constructed to stop water draining back into the site. This cut-off wall will affect the way water moves through the ground all around the site. Two ditches which run through the proposed development site will also need to be combined and rerouted. The changes to water movement could mean that parts of Sizewell Marshes become wetter and some southern parts of the Minsmere South Levels become slightly drier. In the long

term this could lead to changes to habitats in these areas, and changes to the species they support.

2.10 A small population of the rare natterjack toad is also found close to Sizewell, between the South Levels and Sizewell Marshes. The field where their breeding pond is found is currently proposed to become a water management zone during the construction of Sizewell C. Whilst it appears that the breeding pond and hibernation site within the nearly rabbit warren will be avoided, there will still be a significant loss of foraging habitat for the natterjacks. The proposals do not include enough mitigation to enhance connectivity to excellent habitat within Minsmere to the north.

2.11 The proposals for Sizewell C include large amounts of marine infrastructure, including intakes, outfalls, landing facilities and coastal defences. The new structures all have the potential to affect the way coastal processes along the Minsmere and Sizewell frontage work. The coastline in this area is highly dynamic, with climate change and sea level rise adding to the vulnerability of this frontage. We are concerned that some of these structures, particularly the coastal defences, could exacerbate coastal changes in the medium to long term which could affect the future frontage of Minsmere. At present we are concerned that it is not clear what the impacts of this will be.

2.12 The marine environment of the Greater Sizewell Bay is part of a wider marine protected site, important for red-throated divers in winter and for providing foraging areas for Sandwich, common and little tern colonies at Minsmere and other sites nearby. All these species are heavily dependent on their fish prey, and anything which affects the abundance or distribution of their prey species could have knock-on effects on their populations and breeding success. The construction of Sizewell C could affect fish through underwater noise and vibration, increasing suspended sediment though dredging and through discharges of chemicals such as hydrazine during the commissioning stages. During the operation of the power plant, large numbers of fish would be abstracted by the cooling water system. While some of these will be collected and returned to the sea, many of these will already be dead or dying by this point, leading to changes to water quality in the area. This impact could be significantly reduced by the use of an acoustic fish deterrent, in line with best practice and the advice of the Environment Agency, but this is not currently proposed at Sizewell. The return of the cooling water to the sea will also lead to the formation of a thermal plume, which will also contain chemicals such as hydrazine and compounds derived from the chlorine added to the cooling water system. The effects of these on birds and their fish prey is unclear, but of concern, particularly when considered alongside the other impacts affecting the marine environment. The number of vessels required to bring in deliveries to the site is also a cause for concern, particularly in relation to the internationally important red-throated diver population. These birds are highly sensitive to disturbance, with vessel traffic known to cause them to move away from and avoid busy areas.

2.13 The beaches at Sizewell and Minsmere support internationally important vegetated shingle habitats and rare beach nesting birds such as ringed plover. These birds and plants are very vulnerable to human activity on the beaches; the ringed plover are easily flushed from the nests and may abandon them if disturbed significantly or repeatedly by people or dogs while the specialist plants can be easily damaged by trampling. With parts of the footpath network in the Sizewell area being either closed, diverted or just unattractive to users due to the construction works, we are concerned that people are likely to use the neighbouring areas more heavily during the construction period, meaning that sensitive beach wildlife may be affected. In addition, whilst the main visitor areas of the Minsmere reserve have infrastructure designed to allow large numbers of people to visit without disturbing the wildlife, the outer parts of Minsmere are more sensitive and usually much less heavily visited. We are concerned that these areas, particularly the heathlands where the vulnerable stone curlew, woodlark and nightjar nest, may be very attractive to those displaced from the Sizewell area. Without careful management of the impacts

of additional visitors, these heathlands could be damaged by trampling, dog waste and disturbance.

2.14 In summary, we do not believe it will be possible to develop Sizewell C as proposed without significant effects on Minsmere, Sizewell Marshes, Greater Sizewell Bay and the wider nature conservation value of the Suffolk Coast. The proposals have the potential to damage highly protected wildlife sites and to impact nationally and internationally important populations of rare species, which are valued in their own right, by those that live on and visit the Suffolk Coast and all who care about nature conservation across the UK and beyond.

3. ENVIRONMENTAL CONCERNS

Land take from Sizewell Marshes SSSI (including impacts of the SSSI crossing)

Summary

3.1 Sizewell Marshes is designated as a Site of Special Scientific Interest which means it is of national importance and protected under the Wildlife and Countryside Act (1981) as amended.

3.2 The notification[1] for Sizewell Marshes SSSI includes that important for 'large area of lowland, unimproved wet meadows which support outstanding assemblages of invertebrates and breeding birds. Several nationally scarce plants are also present.'[2]

3.3 Sizewell Marshes SSSI is situated to the west of Sizewell nuclear power station and is a complex of marshes with a unique mix of geology and hydrology that has created nationally significant botanical and invertebrate diversity. The site has three main habitats; species rich grazing marsh, tall herb fen and wet alder carr woodland. The value of the SSSI is enhanced by its close proximity and connectivity to RSPB Minsmere.

3.4 Floristically, the grazing marshes and its network of dykes is one of the best examples of this habitat remaining in East Anglia. The dominant vegetation group is M22 *Juncus subnodulosus - Cirsium palustre* fen meadow, with intermediate areas between this dominated by MG10 *Holcus lanatus - Juncus effusus* rush pasture. The M22 *Juncus - Cirsium* communities are very species rich and contain a wide range of plants of conservation interest. The dyke system supports a wealth of plant species including the nationally scarce soft hornwort (*Ceratophyllum submersum*) and whorled water milfoil (*Myriophyllum verticillatum*). The wet woodland W6 *Alnus glutinosa- Urtica dioca* community comprises of chiefly alder and willow with some oak and supports a very high invertebrate diversity.

3.5 18 species of dragonfly have been recorded including rarities such as Norfolk hawker (*Aeshna isosceles*), hairy dragonfly (*Brachytron pratense*) and lesser red eyed damselfly (*Erythromma viridulum*). In total 236 moth species have been recorded since 1996 including *Senta flammea*, a nationally rare species. Further impressive counts include 12 species of grasshopper and cricket, including short winged conehead (*Tetrix subulata*) & Roesel's bush cricket (*Metrioptera roeselli*); a good range of spiders, including the nationally rare *Leiobunum rotundum*; and three nationally rare species of fly: *Odontomyia argentata, Oxycera morrisii* and *Odintomyia ornate.*

3.6 148 bird species have been recorded with a total of 80 breeding species including little egret, gadwall, kingfisher, barn owl & water rail and Cetti's warbler. Wintering birds include bittern, bearded tit and snipe. Mammals include otters, water voles, water shrews and harvest mouse as well as red deer. The SSSI is thought to hold an excellent bat community including barbastelle,

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1 Sizewell Marshes SSSI notification

2 Please refer to [SSSI ecohydrology section] for more details on the nature and conservation value of the interest features within the SSSI.

Natterer's, all three pipistrelle species, brown long eared, noctule, serotine, and Daubenton's recorded annually. It is likely that nationally rare barbastelle and regionally scarce Natterer's bat breed on site.

3.7 Natural England has defined operations likely to damage the special interest[3] of Sizewell Marshes SSSI. Of these, the following are relevant to our concerns:

13a Drainage

13b Modification of the structure of watercourses

14 The changing of water levels and tables and water utilisation

3.8 Natural England views about management[4] for conservation and enhancement of the special conservation interest of wet grassland with breeding and wintering bird interest include the importance of managing water levels and minimising disturbance

- Wet grassland requires active management, traditionally by grazing, if it is to retain its conservation interest. [Periodic hay cutting which allows a breaking up of the thatch, is also important for botanical value.]
- Maintenance of a mosaic of shallow surface pools and un-flooded areas during the winter will provide roosting and feeding habitat for wintering wildfowl and waders
- From April onwards, the area of standing surface water should be reduced to increase the area available for nesting waders and also by concentrating aquatic invertebrates in small pools to provide suitable feeding areas for their young.
- Some shallow areas of flooding should be maintained until late June to provide patches of bare muddy ground on which the birds and their young can feed as raised sward height makes feeding on the drier areas more difficult.
- The birds using these features are directly vulnerable to disturbance, which can cause them to lose time spent feeding or drive them to areas with a poorer supply of food. Management should seek to minimise any harmful disturbance, especially at times when bird populations are under stress, such as severely cold conditions.

3.9 Our main concerns include:

The principle and justification for the proposed loss of part of Sizewell Marshes SSSI

3.10 Vol. 1 ES Addendum Ch. 2 Main Development Site[5] Table 2.38 notes the proposed development will result in the loss of 9.54ha of the SSSI (comprising 6.52ha permanent and 3.02ha 'temporary' loss).

3.11 We are concerned about the principle and scale of the proposed loss of part of Sizewell Marshes SSSI and its justification against the policy tests set out in the Overarching National Policy Statement for Energy (EN-1)[6].

3.12 In addition we are unconvinced about the Applicant's justification for the choice of an embankment and culvert (a single span 'bridge') rather than a triple span bridge to cross Sizewell Marshes SSSI, despite the higher land take from the SSSI. We are concerned that the loss of SSSI

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3 Operations likely to damage the special interest of Sizewell Marshes, Suffolk

4 Views about management of Sizewell Marshes SSSI

5 Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)

6 Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)

via the embankment/culvert option will lead to a much larger loss than the triple span bridge option, and this has not been adequately nor clearly justified.

3.13 Therefore, currently it is our view that the Application has not met the requirement in EN-1 paragraph 5.3.7, namely.

> "*As a general principle, and subject to the specific policies below, development should aim to avoid significant harm to biodiversity and geological conservation interests, including through mitigation and consideration of reasonable alternatives…; where significant harm cannot be avoided, then appropriate compensation measures should be sought*."

3.14 Due to our concerns over the principle of the proposed loss of part of Sizewell Marshes SSSI and its assessment against the tests set out in EN-1.

Temporary land take from Sizewell Marshes SSSI

3.15 We dispute the term 'temporary' damage[7] and we believe it is likely many of the construction activities that will take place (such as repeated tracking across the SSSI) will result in the permanent damage to nationally important fen habitat. We are also concerned about the designation of some loss as 'temporary'[8], which has not been supported by proposals for adequate restoration methods.

Adequacy of the proposed habitat compensation

3.16 Whilst we welcome the Applicant proposals to compensate for the losses of reedbeds and ditches, fen meadow and wet woodland by creating compensatory habitat, we are concerned that the quantity, quality and timing of functionality of all the proposed habitat compensation is inadequate to compensate for loss of all affected SSSI habitats.

Hydrological impacts on Sizewell Marshes SSSI

3.17 Our concerns relating to hydrological impacts on the retained habitats of Sizewell Marshes SSSI are detailed in the SSSI ecohydrology section below.

Introduction

3.18 We are grateful to the Applicant for all the pre-application and pre-examination engagement and discussions. We also engaged with the all consultations (including the Change Application[9]) and repeatedly made our concerns known.

3.19 Therefore it is disappointing to see so many of our concerns remaining. However we remain keen to work further with the Applicant during the Examination process especially on possible management, mitigation and compensation plans as new proposals are brought forward for example the wet woodland strategy[10] submitted at Deadline 1.

3.20 We outline our current position below (as set out in our relevant representations [RR-1059 (RSPB)[11] and RR-1180 (SWT)[12] and our Change Application responses).

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7 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033) 14.7.125
8 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033) 14.7.125
9 ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)
10 9.8 Wet Woodland Strategy (REP1-020)
11 RSPB Relevant Representation (RR-1059)
12 Suffolk Wildlife Trust Relevant Representation (RR-1180)

The principle and justification for the proposed loss of part of Sizewell Marshes SSSI

Introduction

3.21 We are concerned about the principle and scale of the proposed loss of part of Sizewell Marshes SSSI and its justification against the policy tests set out in the Overarching National Policy Statement for Energy (EN-1)[13].

Legislation and Planning Policy

3.22 As set out above we do not agree that the requirement in EN-1[14] paragraph 5.3.7 'to avoid significant harm to biodiversity …through mitigation and the consideration of reasonable alternatives' has been met.

3.23 Although the National Planning Policy Framework[15] (NPPF) (2019) does not contain specific policies for nationally significant infrastructure projects it does note at para 175.

> *When determining planning applications, local planning authorities should apply the following principles: a) if significant harm to biodiversity resulting from a development cannot be avoided (through locating on an alternative site with less harmful impacts), adequately mitigated, or, as a last resort, compensated for, then planning permission should be refused;*
>
> *b) development on land within or outside a Site of Special Scientific Interest, and which is likely to have an adverse effect on it (either individually or in combination with other developments), should not normally be permitted. The only exception is where the benefits of the development in the location proposed clearly outweigh both its likely impact on the features of the site that make it of special scientific interest, and any broader impacts on the national network of Sites of Special Scientific Interest;*

3.24 and at para 170.

> *Planning policies and decisions should contribute to and enhance the natural and local environment by:*
>
> *d) minimising impacts on and providing net gains for biodiversity, including by establishing coherent ecological networks that are more resilient to current and future pressures;*

3.25 This clearly states development *likely to have an adverse effect* on a SSSI should not normally be permitted unless the benefits clearly outweigh its likely impact and that planning decisions should minimise impacts on and provide net gains for biodiversity.

3.26 It is the Applicant view that due to the location of the Application being in the National Policy Statement for Nuclear Power Generation, consideration of alternative locations is not required (ES Non-technical summary, section 3.2[16]).

3.27 The Planning Statement[17] section 8.3.39 notes

> *Therefore, taking into account the overall planning balance set out in Section 11 of this Planning Statement and with particular regard to the measures set out above it can be*

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13 Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)
14 Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)
15 National Planning Policy Framework (2019)
16 Volume 1 6.1 Environmental Statement – Non Technical Summary (APP-159)
17 Planning Statement (APP-590)

concluded that the benefits of the development outweigh the harm and the Sizewell C power station is in accordance with paragraphs 5.3.11, 5.3.14, 5.3.15, 5.3.17 and 5.3.18 of EN-1.

3.28 However in our view, there is inadequate evidence provided to support the conclusion that the *benefits of the development outweigh the harm*. Therefore we consider that the Applicant must provide further evidence to justify its conclusion and that there is no need to consider alternatives.

3.29 EN-1[18] provides more on these important requirements, noting at paragraph 5.3.11.

> *Where a proposed development on land within or outside an SSSI is likely to have an adverse effect on an SSSI….development consent should not normally be granted. Where an adverse effect, after mitigation, on the site's notified special interest features is likely, an exception should only be made where the benefits (including need) of the development at this site, clearly outweigh both the impacts that it is likely to have on the features of the site that make it of special scientific interest and any broader impacts on the national network of SSSIs. … use* [of] *requirements and/or planning obligations* [should be made] *to mitigate the harmful aspects of the development and, where possible, to ensure the conservation and enhancement of the site's biodiversity or geological interest.*

3.30 With the National Policy Statement for Nuclear Power Generation (EN-6)[19] providing more clarity on considerations and requirements specifically in relation to the Application, paragraph C.8.63

> *The Appraisal of Sustainability identified the potential for the mitigation of biodiversity effects on sites of UK wide conservation importance (Sizewell Marshes SSSI), including the creation of replacement habitat. The Appraisal of Sustainability notes that developers could avoid or minimise losses and disturbance to protected species through careful site layout, design, routing, location of the development, associated infrastructure, and construction management and timings. The Appraisal of Sustainability finds that there is potential for habitat creation within the wider area in order to replace lost 'wet meadows' habitats of the Sizewell Marshes SSSI, but also finds that it may not be possible to fully compensate for losses of this habitat. The applicant will need to develop an ecological mitigation and management plan to minimise the impacts.*

SSSI Crossing

3.31 As mentioned above we are unconvinced about the Applicant's justification for the choice of an embankment and culvert rather than a triple span bridge to cross Sizewell Marshes SSSI, despite the higher land take from the SSSI. In addition we are concerned that this larger loss of the SSSI has not been adequately considered nor that the Applicant has minimised all possible impacts to the SSSI and its features.

3.32 The Sustainability Statement[20] submitted with the Application notes at paragraph 3.8.22

> *The crossing would be constructed at the narrowest point to minimise the land-take of the SSSI. The design of the SSSI crossing has evolved to minimise impacts on ecology*.

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18 Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)

19 Department of Energy and Climate Change (2011) National Policy Statement for Nuclear Power Generation (EN-6) Volume II of II - Annexes

20 8.13 Sustainability Statement (APP-617)

3.33 ES Vol. 2 Ch. 6 Alternatives and Design Evolution[21] Table 6.1 compares the SSSI land take of the four SSSI crossing options presented at Stage 2 consultation

Option 1: Causeway over culvert (0.40ha)

Option 2: Single span bridge with vertical wing walls (0.22ha)

Option 3: Three span bridges (0.19ha)

Option 4: Causeway over culvert with adjacent short-term bridge (0.45ha)

3.34 Paragraph 6.2.39 states that the Option 1 causeway over culvert was selected for speed and ease of construction and operational flexibility even though the SSSI land take is considerably higher. APP-190 only assesses the impact of total SSSI loss with the causeway over culvert option selected and claims no residual impact following mitigation without presenting evidence to support this conclusion.

3.35 Nor are the potential impacts of the four SSSI crossing options individually assessed and compared. We consider the impact of all four options should be so assessed and compared to ensure that the right option is chosen to minimise potential effects and losses to the SSSI and its features.

3.36 Noting both the Sustainability Statement above and the Design and Access Statement overarching design principle 11 to

Minimise land take from the SSSI. In this way land take is minimised and where unavoidable, additional compensatory habitat is created. (Part 1[22] , section 6.3.10)

3.37 Along with the High Level Design Principle 10

the development will be designed with the aim of avoiding significant harm to biodiversity (habitats and species) particularly designated interest features of nationally and internationally designated sites, protected and priority species. (In Table 5.1)

3.38 The Application currently clearly does not meet these principles due to its choice of an embankment and culvert.

3.39 Natural England[23] also stated the Applicant would need to demonstrate that the proposals have been designed in a manner that minimises land take from Sizewell Marshes SSSI.

Change to the SSSI crossing design to a single span bridge with embankments

3.40 Whilst the proposed bridge to cross Sizewell Marshes SSSI in the Change Application (Vol. 1 ES Addendum Ch. 2 Main Development Site[24]) appears to be an improvement on the plans presented in the Application (ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[25]) (from an ecological standpoint), we still consider that the bridge should be designed more sensitively as a three span bridge to further reduce SSSI loss from the crossing and to provide greater connectivity for species/groups including invertebrates, water voles, otters and bats, thereby reducing the potential for fragmentation of populations.

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21 ES Volume 2 Main Development Site Chapter 6 Description of Alternatives and Design Evolution (APP-190)
22 Main Development Site Design and Access Statement Part 1 (APP-585)
23 Natural England Relevant Representation [RR-0878] 103.
24 ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)
25 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033)

3.41 We note the commitment in Vol. 1 ES Addendum Ch. 2 Main Development Site[26] paragraph 2.2.138 to the provision of a ledge to allow otters to pass and the incorporation of bat roosts into the structure, however, detailed designs and full assessment of the potential impacts from the proposed bridge over the Sizewell Marshes SSSI are required.

3.42 We also note that whilst the revised design may improve connectivity for some species (e.g. otters) compared to the design in the Application, some species (in particular, certain species of invertebrates) will still be significantly negatively affected by the proposals. Again, our preference would be for a three-span bridge which would result in higher light levels reaching the watercourse below and hence greater connectivity for all affected species.

3.43 Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[27] paragraph 2.9.65 notes that the revised design will reduce land-take from Sizewell Marshes SSSI by 0.08ha. Whilst this intention is welcomed, the sections of SSSI remaining in the bridge section would be impacted during the construction period, potentially over a continuous period of a number of years and more evidence is required to demonstrate how potential harm can be minimised as well as potential for their restoration, despite being in such close proximity to the causeway structure (discussed further below).

3.44 A detailed design for the SSSI crossing has not been submitted to the Examination. ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[28] notes at paragraph 14.4.10

> *The final dimensions would be confirmed as the design progresses*.

3.45 This means the impacts of the crossing cannot be properly assessed or scrutinised by Interested Parties and the Examination Authority.

3.46 This does not comply with EN-1[29] which notes at paragraph 4.2.8

> *Where some details are still to be finalised the ES should set out, to the best of the applicant's knowledge, what the maximum extent of the proposed development may be in terms of site and plant specifications, and assess, on that basis, the effects which the project could have to ensure that the impacts of the project as it may be constructed have been properly assessed.*

3.47 EN-6[30] goes on to note, paragraph C2.51

> *The potential impacts of development on designated habitats will be taken into account in the project level assessments and considered by the* [PINs] *as part of the application for development consent.*

3.48 EN-6 notes at paragraph C.8.67

> *The* [PINs] *should also refer to the Appraisal of Sustainability … and consider whether the applicant's proposals have sufficiently taken into account the issues identified, where they are still relevant.*

3.49 We are concerned that the information before the Examination is inadequate for the Examining Authority to consider fully and robustly all possible effects on the SSSI and its features.

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[26] ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)
[27] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033)
[28] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033)
[29] Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)
[30] Department of Energy and Climate Change (2011) National Policy Statement for Nuclear Power Generation (EN-6) Volume II of II - Annexes

Temporary land take from Sizewell Marshes SSSI

3.50 ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[31] paragraph 14.7.125 notes temporary loss includes habitats that would be impacted temporarily during the construction phase and those where replacement habitat is proposed within the site. Table 14.10 shows the temporary land take from wet woodland, fen meadow, wet reedbed and ditches. 14.7.125 explains the area of temporary impact includes wet woodland and fen meadow located within the corridor which will be used to install overhead power lines and notes

> *The wet woodland would be coppiced to enable the cable to be laid out, prior to lifting and the fen meadow would be protected from damage using appropriate methods for spreading the weight of plant in wet ground.*

3.51 We dispute the term 'temporary' impact and we believe it is likely many of the activities that will take place (such as repeated tracking across the SSSI) will result in the permanent damage to nationally important fen habitat. The Applicant has not presented evidence to demonstrate how the fen meadow will be protected from damage and how the habitat will be able to be re-established after construction. In our view the Applicant should provide detailed evidence for scrutiny during the Examination.

3.52 Details to support the temporary loss of wet reedbed and ditches are missing and we request the Applicant provides evidence to support the assertion of temporary loss of those habitats.

3.53 We are also concerned about the designation of some loss as 'temporary', because it has not been supported by proposals for adequate restoration methods.

3.54 Vol. 1 ES Addendum Ch. 2 Main Development Site[32] Table 2.36: Revised Sizewell Marshes SSSI land take calculations show 0.61ha temporary and 0.46ha permanent loss of fen meadow and 2.23ha temporary and 3.06ha permanent loss of wet woodland.

3.55 We have significant concern that areas of 'temporary' fen meadow loss will suffer long term damage and hence there could be a total of 1.07ha of permanent loss and permanent damage. We request monitoring of the condition of the fen meadow subject to 'temporary' loss and if a significant effect is identified, further habitat creation would be required to compensate for this.

3.56 The 'temporary' loss is largely wet woodland which will take a long time to return to the habitat quality at the start of the 'temporary' land take.

3.57 Table 6.1 of the outline Landscape and Ecological Management Plan (oLEMP)[33] notes the proposed management for habitat reinstatement of Sizewell Marshes SSSI Marsh, Fen and Reedbed

> *The area of marsh, fen and reedbed habitats within the temporary construction area for the SSSI crossing in the Sizewell Marshes SSSI would be reinstated following construction completion. A detailed re-instatement plan would be included within the detailed LEMP.*

3.58 But it does not mention a re-instatement plan for temporary loss of wet woodland.

3.59 We request the Applicant submits a detailed habitat re-instatement plan for all habitats subject to 'temporary' loss for scrutiny during the Examination process.

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31 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033)

32 ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)

33 8.2 Outline Landscape and Ecological Management Plan (oLEMP) (REP1-010)

Adequacy of the proposed habitat compensation

3.60 ES Vol 2 Ch 14 Terrestrial Ecology and Ornithology[34] Table 14.12 summarises the effects from the construction phase on Sizewell Marshes SSSI. It concludes a moderate adverse effect from direct land take resulting in habitat loss and fragmentation would be reduced to minor adverse (not significant) with additional mitigation. The additional mitigation proposed includes

> *Habitat monitoring to ensure habitat creation meets its objectives. Aim to create/ restore more fen meadow habitat than is being lost to allow for uncertainties in creation; use of two sites reduces risk Compensatory funding in the event of failure to deliver compensatory habitats at either location, to enable fen meadow creation at other sites. Wet woodland strategy to identify further wet woodland habitat opportunities*.

3.61 Vol. 1 ES Addendum Ch. 2 Main Development Site[35] submitted with the change Application concludes impacts to Sizewell Marshes SSSI are considered to remain as minor adverse (not significant).

3.62 The Applicant proposes to compensate for the losses of reedbeds and ditches, fen meadow and wet woodland by creating compensatory habitat. We request the Applicant provides a plan with all proposed compensation areas for all affected habitats clearly annotated.

Quantity, quality and timing of functionality of all the proposed habitat compensation

3.63 We are concerned that the quantity, quality and timing of functionality of all the proposed habitat compensation is inadequate to compensate for loss of all affected SSSI habitats.

3.64 For example Aldhurst Farm habitat creation[36] has been designed to compensate for the loss of reedbed and ditch habitat only. We are keen to see that any habitat creation is of the highest quality and achieves the best possible outcomes for wildlife (irrespective of whether this becomes an enhancement or compensation site). Currently we do not agree that it will compensate for the SSSI loss since this continues to be inadequately demonstrated.

3.65 We are also concerned about the lack of an attempt to address loss of fen meadow, wet woodland and specialist invertebrates. We therefore consider there will be residual impacts (other than reedbed) and we are not confident that compensation more recently proposed will fully address this.

3.66 Compensation habitat should be in place and functional before habitat loss occurs. The construction programme should follow the appropriate compensation work and not the other way around. We request the Applicant defines the seasons when the habitat creation work would need to take place. The timing of the construction programme needs to accommodate this.

Fen meadow

3.67 As stated above the revised Sizewell Marshes SSSI land take calculations in Vol. 1 ES Addendum Ch. 2 Main Development Site[37] show 0.61ha temporary and 0.46ha permanent loss of fen meadow.

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34 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033)
35 ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181) 2.9.52, 2.9.65, 2.9.86
36 ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181) 14.4.11
37 ES Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)

3.68 The Fen Meadow Strategy (Vol. 3 Ch. 2 ES Addendum Terrestrial Ecology and Ornithology Appendices 2.9.A-2.9.D Part 2[38]) proposes creation of 4.5 ha of fen meadow at three sites (Benhall, Halesworth and Pakenham) to compensate for the permanent loss of approximately 0.5ha of fen meadow within the SSSI.

3.69 Whilst the provision of more SSSI compensatory habitat (paragraph 1.1.3) is welcome, the distance from the area of loss at Sizewell Marshes SSSI is of concern. Compensation sites should be as close to the lost habitats as possible and given the distance of Pakenham from Sizewell, there is a clear limitation of Pakenham in terms of compensation for the loss of the Sizewell Marshes SSSI. Consideration should be given to increasing the compensation ratio to take account of the distance of the compensation site from Sizewell Marshes SSSI.

3.70 Although the EC guidance on "Managing Natura 2000 sites" (EC, 2018)[39] is not directly relevant to SSSIs it does helpfully explain that the extent of compensation required is directly related to:

- the quantitative and qualitative aspects inherent to the elements of integrity likely to be impaired
- the estimated effectiveness of the measure(s) [section 5.5.4]

3.71 The EC Guidance (2018) also states that ratios are best set on a case-by case basis, but should generally be well above 1:1. Ratios of 1:1 or below should only be considered when it can be shown that measures will be fully effective in reinstating structure and functionality in a short period of time [section 5.5.4].

3.72 Paragraph 5.1.8 of the Strategy notes:

> *Once the land for the three fen meadow locations has been secured, either by agreement or by compulsory acquisition powers in the DCO as made, SZC Co will undertake the relevant 'Year 1 works' to be defined in the Fen Meadow Plan and as detailed below at 5.1.9 onwards, within a period to be defined within the plan and linked to the commencement of works in the DCO as made.*

3.73 The Strategy does not mention when the habitat creation works will commence in relation to the construction programme and SSSI land-take. Section 6 states the target is to establish at least 4.5ha of M22 fen meadow habitat across the three fen meadow sites within 10 years of commencing the habitat creation works. The Indicative Phasing Schedule shows fen meadow habitat creation during construction years 1 to 3. Compensation habitat should be in place and functional before habitat loss occurs. The construction programme should follow the appropriate compensation work and not the other way around. We request the Applicant defines the seasons when the habitat creation work would need to take place. The timing of the construction programme needs to accommodate this.

3.74 Due to potentially high nutrient levels found in the surface water, the newly proposed fen compensation sites may only support more generalist species, with a resulting overall loss of biodiversity. We have seen no evidence as yet regarding the technical feasibility of fen meadow creation on these sites.

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[38] Volume 3 Chapter 2 Environmental Statement Addendum Terrestrial Ecology and Ornithology Appendices 2.9.A-2.9.D Part 2 (AS-209)

[39] European Commission (2018) Managing Natura 2000 sites. The provisions of Article 6 of the 'Habitats' Directive 92/43/EEC section 5.5.4

3.75 We request the Applicant provides peer-reviewed evidence from similar schemes to demonstrate it is feasible to successfully create M22 species-rich fen/fen meadow habitat. We also request clarification of the next steps if it is found that it is not feasible to create this habitat.

3.76 Paragraph 1.1.4 proposes that a 'Fen Meadow Plan' be prepared in accordance with the Fen Meadow Strategy and be subject to a DCO Requirement 14A.

3.77 We are concerned that the detailed surveys to inform the Fen Meadow Plan (4.1.9) will not be completed until after the close of the Examination. We are still awaiting the Fen Meadow Plan Report 1 which the fen meadow strategy (4.1.11) noted '*will be prepared for submission in Q1 2021, to provide the baseline reports for the sites and water data available to that period, timed to inform stakeholders and the commencement of the examination*'.

3.78 If the detailed surveys do not provide confidence that fen meadow establishment is achievable or there is a high degree of uncertainty, then other sites should be identified to deliver new fen meadow habitats.

3.79 Paragraph 4.1.11 also notes The Fen Meadow Plan Draft 1 to be submitted later in the examination process '*will provide further interim data and define in draft the management interventions required to create fen meadow habitats'*. We welcome the commitment to include draft monitoring and remedial measures to ensure SSSI quality fen meadow is delivered. The delivery of fen meadow compensation must be legally and financially secured via the S.106 Agreement with flexibility to take account of the fact that the Plan will be finalised after the close of the Examination.

3.80 With regard to the plans to transfer green hay to the Pakenham site discussed in paragraph 5.1.9 and 5.1.11, in terms of ecological best practice these transfers should be from the Pakenham Meadows SSSI.

3.81 We welcome the commitment to include surveys of existing fen meadow local to the compensation site, including Pakenham Meadows SSSI, to identify species (e.g. invertebrates) present locally and then target measures to encourage their colonisation of the newly created fen meadow habitat (4.1.10).

Potential impacts from the proposed Pakenham Fen site on Pakenham Meadows SSSI

3.82 The potential impacts from the proposed Pakenham Fen site on the features of the adjacent Pakenham Meadows SSSI need to be assessed. Ecological, groundwater and surface water surveys of the Pakenham Meadows SSSI should be undertaken to establish its condition before construction of the Pakenham compensation area. The SSSI should be monitored during and after construction to detect any potential impacts from groundwater and surface water changes on its features, allowing rapid deployment of mitigation measures should any changes occur.

Potential impacts from the Benhall Fen Meadow site on the Abbey Farm compensation site

3.83 We are concerned about potential impacts from the Benhall Fen Meadow site on the Abbey Farm compensation site that is providing compensatory habitat for expected losses of part of the coastal reedbeds to sea level rise and managed realignment[40] [41] – specifically to compensate for

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[40] Suffolk Coastal District Council, Waveney District Council and the Environment Agency (2010) Anglian Region Habitat Creation Programme http://www.suffolksmp2.org.uk/publicdocuments/finalsmp2/Regional%20Habitats%20Creation%20Programme.doc Section 1.2, Table 1, Table 2, Table 3

[41] Shoreline Management Plan 7 (suffolksmp2.org.uk)

adverse effects on the Benacre to Easton Bavents Special Protection Area (SPA), and the Minsmere-Walberswick SPA and Ramsar site.

3.84 The Benhall Fen Meadow site is on the River Fromus catchment, upstream of the Abbey Farm compensation site, where the RSPB has been managing the compensatory habitat[42]. Abbey Farm is functioning as reedbed now and supports breeding bittern and marsh harrier. The site manager (pers. comms.) confirmed the site is currently vulnerable to water supply when river levels drop in the summer months.

3.85 If the Applicant needs to hold more water back upstream in the summer for the Benhall fen meadow compensation site this could exacerbate the issues downstream at the Abbey Farm compensation site.

3.86 In addition it is important to note that it is government policy) that sites identified, or required, as compensatory measures for *adverse effects* on protected sites (SPAs, SACs and Ramsar sites) should be given the same protection as those protected sites[43], with EC Guidance (2018) also requiring that they themselves become protected sites[44] . Therefore, consideration of potential effects of the proposed Benhall compensation site on the Abbey Farm compensation site should be undertaken and should *likely significant effect* be identified, an HRA carried out. We request the Applicant provides modelling and a thorough assessment of the potential impacts of the proposed Benhall compensation site on the Abbey Farm compensation site.

Wet woodland

3.87 The Wet Woodland Strategy[45] (paragraph 1.1.2) proposes creation of 3.06ha of wet woodland, with 0.7ha to be created on the EDF estate and 2.36ha created between the Benhall and Pakenham fen meadow compensation sites. Since compensation sites should be as close as possible to the lost habitats, we consider the compensation ratio should be increased to take account of the distance of the off-site compensation sites from Sizewell Marshes SSSI but without constraining the creation of fen meadow habitat.

3.88 Section 6 of the Strategy notes establishment of the wet woodland habitats would commence within 12 months of the commencement of development under the DCO and continue for 10 years. The test of success (paragraph 7.1.1 of the Strategy) is the establishment by Year 10, of at least 3.06ha of developing wet woodland habitat. Compensation habitat should be functional before habitat loss occurs. We are concerned it will not be functional for at least 10 years following loss of the SSSI habitat

3.89 Paragraph 5.1.10 of the Strategy also contains a proposal to create 0.7ha wet woodland habitat in a new wetland corridor within the marsh harrier habitat improvement area. We are concerned that the creation of wet woodland within the marsh harrier compensation area will result in a loss of habitat available for foraging marsh harriers (as discussed below) as the habitat is not likely to be functional by the time construction starts.

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42 Suffolk Coastal District Council, Waveney District Council and the Environment Agency (2010) Anglian Region Habitat Creation Programme http://www.suffolksmp2.org.uk/publicdocuments/finalsmp2/Regional%20Habitats%20Creation%20Programme.doc Table 4 (Snape, Suffolk)

43 Ministry of Housing, Communities and Local Government (2019) National Planning Policy Framework (NPPF), paragraph 176

44 European Commission (2018) Managing Natura 2000 sites. The provisions of Article 6 of the 'Habitats' Directive 92/43/EEC section 5.5.5

45 9.8 Wet Woodland Strategy (REP1-020)

3.90 Para 8.1.2 of the Strategy refers to contingency approach to deliver the balance of wet woodland and any additional required to meet the total of 3.0ha on the EDF Energy estate at either Aldhurst Farm or on the new wetlands which would have been established on the marsh harrier compensatory habitat area in the north-east of the EDF Energy estate. We consider compensation at Aldhurst Farm should focus on reedbed and wet woodland should not be created there at the expense of reedbed.

3.91 Due to the many concerns above about adequacy, we strongly recommend that the need for contingency habitat is considered now and information submitted to the Examination, so that it can be taken account of by the ExA. Also, there will be a need to consider potential impacts on neighbouring habitats e.g. if waders are nesting nearby it is not ideal to introduce tree cover which provides perching for predators.

Conclusions

3.92 In conclusion, based on the information available at present, our key concerns are

- The principle and scale of the proposed loss of part of Sizewell Marshes SSSI and its justification against the policy tests set out in the Overarching National Policy Statement for Energy (EN-1)
- The choice of an embankment and culvert (a single span 'bridge') rather than a triple span bridge to cross Sizewell Marshes SSSI, despite the higher land take from the SSSI, has not been adequately and clearly justified
- We consider the Applicant should assess and compare the impact of all four options for the SSSI crossing and provide sufficient evidence for Natural England to advise the Examination on any effects on the special interest of the SSSI.
- A detailed design for the SSSI crossing has not been submitted to the Examination.
- We do not agree that the Application has met the requirement in EN-1 paragraph 5.3.7

  to avoid significant harm to biodiversity …through mitigation and the consideration of reasonable alternatives

- We dispute the term 'temporary' impact and we believe it is likely many of the activities that will take place will result in permanent damage to nationally important fen habitat
- The designation of some loss as 'temporary' has not been supported by a detailed habitat re-instatement plan for all habitats subject to 'temporary' loss
- The quantity, quality and timing of functionality of all the proposed habitat compensation is inadequate to compensate for loss of all affected SSSI habitats
- Compensation habitat should be in place and functional before habitat loss occurs. The construction programme should follow the appropriate compensation work and not the other way around. We request the Applicant defines the seasons when the habitat creation work would need to take place. The timing of the construction programme needs to accommodate this.
- We request the Applicant provides peer-reviewed evidence from similar schemes to demonstrate it is feasible to successfully create fen meadow habitat. We also request clarification of the next steps if it is found that it is not feasible to create this habitat.
- We request the Applicant provides modelling and a thorough assessment of the potential impacts of the proposed Benhall compensation site on the Abbey Farm compensation site.
- Due to the many concerns about adequacy of compensation habitat, we strongly recommend that the need for contingency habitat is considered now and information submitted to the Examination, so that it can be taken account of by the ExA.

3.93 More information and details are required now for these proposals to be fully considered and taken account of.

Coastal Processes and RSPB Minsmere

Introduction

3.94 The RSPB owns the coastal frontage running from the boundary with the EDF estate (just north of the Application red line boundary) to the boundary with National Trust, north of the Minsmere Sluice, where the Coney Hill cross bank/north wall meets the coast (at approximately OS Grid Ref TM478671).





RSPB Minsmere shoreline, looking north from Minsmere Sluice towards Dunwich Cliffs. Copyright Tormod Amundsen/Biotope

3.95 The coastal frontage forms part of the Minsmere to Walberswick SAC, SPA, Ramsar site and SSSI (the designated sites) and is particularly important for its vegetated shingle and coastal dune habitat, the former of international importance for the annual vegetation of driftlines[46] and perennial vegetation of stony banks[47] floral communities.

3.96 The annual vegetation of driftlines feature is a primary reason for site selection of the Minsmere – Walberswick SAC, represented on the Minsmere frontage by the presence of orache *Atriplex* spp and sea sandwort *Honckenya peploides.* The site is one of only two representative sites of this habitat on the east coast of England and one of only four known outstanding localities in the United Kingdom. This habitat is considered to be rare as its total extent in the UK is estimated to be less than 100ha. The Perennial vegetation of stony banks is a qualifying feature of the Minsmere – Walberswick SAC, represented on the Minsmere frontage by the presence of the yellow horned-poppy *Glaucium flavum* and the rare sea-kale *Crambe maritima* and sea pea *Lathyrus japonicus.* The last two species are also qualifying features of the Ramsar designation and all three species are cited in the SSSI designation.

46 Annual vegetation of drift lines (Annual vegetation of drift lines) - Special Areas of Conservation (jncc.gov.uk)

47 Coastal shingle vegetation outside the reach of waves (Perennial vegetation of stony banks) - Special Areas of Conservation (jncc.gov.uk)

3.97 A number of studies (eg Cadbury 199648, 200449, 201550) have confirmed the importance of the vegetation on this shingle frontage. The shingle and dune barrier also provide protection for the nationally and internationally important brackish and freshwater habitats on the landward side, which form valuable low-lying habitat in the Minsmere valley, part of the Minsmere to Walberswick SPA, SAC, Ramsar and SSSI sites and managed by the RSPB as part of Minsmere reserve.

3.98 This stretch of the coast is a dynamic system and it can be difficult to interpret or predict natural geomorphological change. However, Pye & Blott (2005)[51], in response to an Environment Agency consultation, concluded that there was no scientific basis to conclude that a major coastal realignment, involving virtual abandonment of the freshwater habitat interests at Minsmere, is required in the short to medium term (i.e in the next 50 years). This was also the conclusion of the Minsmere Flood Risk Management Study (Environment Agency 2009)[52], which concluded that the defences south of the Minsmere Sluice were less likely to be at risk of coastal erosion over the next 100 years. This study informed the Environment Agency's Minsmere Sea Defences project, completed in 2012, that afforded protection to freshwater habitats (Schofield & Ferguson 2014)[53].

3.99 Studies used to inform EA's project indicated that the enhancements to the Coney Hill cross bank and adjacent areas should allow the freshwater habitats to be maintained *in situ* for the next 50 years at least. The studies (Black & Veatch 2006[54]) proposed that the majority of the RSPB Minsmere frontage had a residual life of between 50 – 200 years. Therefore, but for the Application, there is no reason to believe additional defences will be needed in the medium term, nor that the designated sites and their features are at risk during this period also.

Sites and species of concern

3.100 As described in more detail above, the following are the protected sites, species and other protections relevant to this issue:

- Minsmere to Walberswick SAC, SPA, Ramsar, SSSI

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48 Cadbury, C. J. (1996) Vegetation survey of the Coastal Grazing Marshes and Dunes at Minsmere RSPB Nature Reserve, 1993. *Unpubl. rept. Reserves Ecology.* RSPB, Sandy

49 Cadbury, C. J. (2004) Repeat vegetation survey of the Beach and Dunes at Minsmere RSPB Nature Reserve, Suffolk, 2004. *Unpubl. rept. Reserves Ecology.* RSPB, Sandy

50 Cadbury, C.J. (2015) Assessing the Impact of the Tidal Surge of 5/6 December 2013 on the Shore Profile and Flora of RSPB Nature Reserves on the Norfolk and Suffolk coast. *Unpubl. rept. Reserves Ecology.* RSPB, Sandy

51 Pye, Prof. K. & Blott, Dr. S. (2005) Coastal Processes and Morphological Evolution of the Minsmere Reserve and Surrounding Area, Suffolk. *Unpubl. Report for RSPB* - This was an RSPB-commissioned study in response to an EA consultation on future sea defences for Minsmere that included all options from hold the line to full retreat

52 Environment Agency (2009) Planning a sustainable approach: Minsmere Flood Risk Management Study

53 Schofield, A. & Ferguson, E. (2014) Minsmere Flood Risk Management Scheme. *UK Water Projects 2013-2014 – Virtual Edition*

54 Black & Veatch (2006) Timings and Longevity Predictions for Minsmere. *Non-technical Summary. Report for Environment Agency*

- RSPB Minsmere Council of Europe Diploma site – in recognition of the maintenance of an exceptional European interest with exemplary management (Council of Europe[55] 1979 and 2019[56])[57]
- Annual vegetation of drift lines (Annex 1 SAC qualifying feature and primary reason for site designation)
- Perennial vegetation of stony banks (Annex 1 SAC qualifying feature)
- Wetland bird populations threatened by flooding of freshwater habitat (including shoveler, gadwall, bittern, marsh harrier, avocet & little tern)
- Breeding ringed plover
- Sizewell beach County Wildlife Site (identified in Doody & Randall 2003[58] as being an important site for coastal vegetated shingle)
- Heritage Coast[59]
- Invertebrate communities on shingle - the shingle invertebrate assemblage on the Minsmere frontage has not been systematically surveyed but is likely to support an important invertebrate community.

International Importance

3.101 Doody & Randall (2003) identify that Great Britain supports a significant proportion of European shingle shore structures and that vegetated shingle (comprised of Annual vegetation of driftlines and Perennial vegetation of stony banks) is recognised as an internationally important habitat, particularly as it is scarce in the rest of Europe with the UK supporting a high proportion of the European resource. Elsewhere globally, coastal shingle has few occurrences outside of Japan and New Zealand. The vegetated shingle on the Suffolk Coast is therefore of significant global importance and is one of the main reasons it is designated as a SAC and Ramsar site.

Summary of concerns relating to coastal processes

3.102 As described in ES *Vol. 2 Ch. 2 Description of Permanent Development*[60] and *ESVol. 2 Ch. 20 Coastal Geomorphology and Hydrodynamics*[61], various aspects of the proposed development have the potential to interfere with the coastal processes on the Minsmere frontage. These include:

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55 Council of Europe (1979) Resolution (79)13. On the award of the European Diploma to the Minsmere nature reserve (United Kingdom)

56 Council of Europe (2019) Resolution CM/ResDip(2019)5 on the renewal of the European Diploma for Protected Areas awarded to the Minsmere Nature Reserve (United Kingdom)

57 The European Diploma for Protected Areas is a prestigious international award for **the efficient protection and management of landscapes, reserves or natural monuments and sites,** granted since 1965 by the Committee of Ministers of the Council of Europe. It recognises natural and semi-natural areas and landscapes of exceptional European importance for the preservation of biological, geological and landscape diversity which are managed in an exemplary way. Protected areas can receive the Diploma due to outstanding scientific, cultural or aesthetic qualities, but they must also have suitable conservation management. Since its creation in 1965, 74 protected areas, in 29 countries (both EU and non EU) have been granted the European Diploma. Minsmere is one of only five sites in the UK to have achieved this award.

58 Doody, Dr P. & Randall, Dr R. (2003) *A Guide to the Management and Restoration of Coastal Vegetated Shingle.* English Nature report

59 The national policy framework and objectives for heritage coasts were developed by the Countryside Commission, a predecessor of Natural England, and ratified by government. Heritage coasts: definition, purpose and Natural England's role - GOV.UK (www.gov.uk) Heritage Coast is a definition to conserve the best stretches of undeveloped coast in England, established to conserve protect and enhance the natural beauty of the coastline; the terrestrial coastal and marine flora and fauna and their heritage features (see Natural England heritage coast definition and purpose – accessed 27 May 2021)

60 ES Volume 2 Main Development Site Chapter 2 Description of Permanent Development (APP-180)

61 ES Volume 2 Main Development Site Chapter 20 Coastal Geomorphology and Hydrodynamics (APP-311)

- The Hard Coastal Defence Feature (HCDF)[62]
- The Soft Coastal Defence Feature (SCDF) [63]
- The permanent beach landing facility (BLF) and temporary BLF, introduced in the Change Application)[64]
- Two Fish Return and Recovery Outfalls (FRR)[65] and Combined Drainage Outfall (CDO) [66]

3.103 The potential for these structures and associated activities to affect coastal processes on the shoreline have been identified in the Environmental Statement and HRA accompanying the Application.

3.104 Risk of potential impacts to the Minsmere – Walberswick designated SAC, SPA, Ramsar and SSSI sites include:

- Accelerated erosion which could:
  - Lead to the loss of habitat supporting SAC and SSSI designated flora
  - Accelerate a breach of the shingle and dune barrier bank, allowing saline incursion and flooding into the brackish and freshwater areas of RSPB Minsmere (affecting the SPA and Ramsar site species) and Sizewell Marshes SSSI and any breach of the barrier bank has the potential to impact on the visitor infrastructure on the RSPB Minsmere reserve
  - Affect access routes for the public, potentially increasing trampling and disturbance impacts on the SAC and SPA features due to alternative routes being sought
  - Affect nesting bird species e.g. little tern (SPA feature) and ringed plover through loss of habitat and/or increased disturbance

- Increased accretion
  - Leading to a change in coastal dynamics and coastal composition which could affect the SAC, Ramsar and SSSI designated flora
  - This could provide increased protection for the brackish and freshwater habitats and visitor infrastructure at RSPB Minsmere, but would require management that was sympathetic to the potential adverse impact[67]

3.105 Section 20.4 of ES*Vol. 2 Ch. 20 Coastal Geomorphology and Hydrodynamics* describes the baseline geomorphology of the Greater Sizewell Bay, including the shoreline, nearshore and offshore bars and Sizewell to Dunwich offshore bank.

3.106 Should the two FRR and CDO outfalls have an effect on the nearshore bars, this could affect sediment supply or lead to more storm damage to the adjacent sections of coast, potentially increasing erosion effects. Pye & Blott (2005)[68] amongst others have illustrated the apparent relationship between the nearshore and offshore bars and patterns of local erosion and accretion along the shoreline at Minsmere and surrounding areas and have highlighted that changes to the nearshore and offshore bars appear to be related to episodes of erosion or accretion along the shoreline.

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62 As described within APP-180: section 2.4.57 to 2.4.60 pp16 & 17; APP311 section 2.5 pp29 – 32

63 As described within APP-180: section 2.4.58 p 17; APP311 section 2.5 pp29 – 32

64 As described within APP-180: section 2.4.66 to 2.4.75 pp17 & 18; APP311 section 2.5 pp29 – 32

65 The FRR outfalls enable larger organisms caught on the drum screens during the abstraction of the cooling water to be returned to the sea.

66 As described within APP-180: section 2.6.7 to 2.6.9 p 31; APP311 section 2.5 pp29 – 32

67 Increased deposition of sediment on the foreshore, which could happen as a consequence of the SZC offshore infrastructure influencing sediment transport, would cause an accumulation of sand and shingle, building the beach and providing increased protection to Minsmere

68 Pye, Prof. K. & Blott, Dr. S. (2005) Coastal Processes and Morphological Evolution of the Minsmere Reserve and Surrounding Area, Suffolk. *Unpubl. Report for RSPB*

3.107 Little tern (SPA feature) nested on the coastal shingle at Minsmere until 2008 and have nested on the coastal lagoons in more recent years. The species may return to nest on the coastal shingle in future years and needs to be considered as a potential breeding species. Suitable sand and shingle habitat, with reduced human disturbance is essential to attract this species to settle and breed successfully. Ringed plover (not a designated feature, but of local importance) continue to breed annually on the coastal shingle shoreline, including the area south of Minsmere Sluice and again rely on suitable sand and shingle habitat and limited disturbance.

3.108 Doody and Randall (2003)[69] highlight that vegetated coastal shingle in the UK is largely protected as SSSI, Heritage Coast, AONB and ownership by the RSPB and others, which should allow protection from damage by activities such as coastal defence works. They note that man's intervention in natural coastal processes represents one of the main long-term threats to vegetated shingle, with coast protection work changing the accretion rate of shingle to coastal areas. They acknowledge where coastal protection techniques involve a more integrated approach allowing natural processes to work where possible and relying more on 'soft' defences rather than 'hard' structures such as sea walls, there is an opportunity to create new, stable areas of shingle. Although they recognise that if new stable areas of shingle are created these could also create new areas of shingle vegetation, they note it does not always follow that this new area of shingle will restore the plant and animal assemblages of areas that have been lost.

3.109 ES *Vol. 2 Ch. 20 Coastal Geomorphology and Hydrodynamics* section 20.7.4 page 32 describes initial investigations suggesting shingle won from the excavation of the footings for the HCDF will be of suitable size and quality to be used as source material for construction of the SCDF.

3.110 However it is noted that this is subject to a further suitability assessment once excavations begin; otherwise sediments for the SCDF would be delivered to the site (from a licenced aggregate extraction site and/or using excavated material from the main development site) rather than reprofiling the beach. *Vol. 3: ES Addendum Appendices Ch. 2 Main Development Site Appendix 2.15.A Coastal Geomorphology and Hydrodynamics*[70] *section 6.5 page 33* confirms that mitigation methods for the SCDF would involve either moving existing beach sediment (bypassing or recycling) or introducing new material (recharge).

3.111 Doody and Randall (2003) note that ongoing beach recycling operations relating to the introduction of sea defence structures (e.g. sea walls) often involve activities which have a significant impact on shingle vegetation. The act of moving the beach material does have a profound impact on the vegetation and any invertebrates associated with stable or semi stable shingle structures. Problems can occur where long-term works are carried out as this will prevent any natural beach profiles (or reprofiling following disruption) being allowed to develop.

3.112 Beach recharge is often seen as a more environmentally friendly 'softer' option for coastal defence. However, the import of material foreign to the locality can bring with it its own pressures and issues. Imported marine-dredged shingle can cause nutrient enrichment (or possibly depletion) and adverse effects on local beaches can occur when shingle of unsuitable type/size is used.

3.113 Alternatively, beach recycling prevents the accretion of 'pioneer' shingle ridges[71], essential for the SAC Annex 1 annual vegetation of drift lines feature and a key component of the mosaic of

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69 Doody, Dr P. & Randall, Dr R. (2003) *A Guide to the Management and Restoration of Coastal Vegetated Shingle.* English Nature report

70 Volume 3: Environmental Statement Addendum Appendices Chapter 2 Main Development Site Appendix 2.15.A Coastal Geomorphology and Hydrodynamics (AS-237)

71 Pioneer shingle ridges are new, naturally formed shingle ridges, created by natural coastal processes that in turn provide habitat for the vegetation communities

habitats for scarce invertebrates (part of the Ramsar site listing), due to the regular intervention to move beach material from one location to another. In summary, Doody and Randall (2003)[72] consider beach nourishment can be detrimental in increasing the amount of fine sediment within the shingle, hence changing the conditions required by pioneer species (such as *Atriplex* spp, sea kale and sea pea). They note that if the new or replenished beach is allowed to develop without interference through reprofiling and the material is washed to reduce nutrient enrichment, the process can provide the foundation for restoration of vegetation (and the species reliant on that vegetation).

3.114 However, in many instances this is not the case and continued intervention and management of beach material to increase crest height and 'improve' the sea defence capability prevents such colonisation, confirming that it therefore represents ongoing negative impacts on many designated sites and their species.

3.115 Doody and Randall (2003) also note that the most damaging activity affecting coastal vegetated shingle is the removal of gravel from the beach itself. This action destroys the vegetation and associated fauna which, due to the complex structure in which these communities develop, is impossible to recreate.

3.116 Therefore, Doody and Randall 2003 question the appropriateness of coastal defence structures and beach reprofiling/nourishment within important nature conservation sites. Sites with a dynamic balance between stability and instability allow for a variety of communities to develop, including those associated with both ephemeral and more stable structures.

3.117 The designated Minsmere to Walberswick SAC, Ramsar and SSSI sites have fixed boundaries, with lines on maps used to define the protected areas. This can impose a rigidity on restoration activities, which is not commensurate with the inherent dynamism of the systems. This is particularly important where shingle beaches or structures move in response to tidal energy and storms, beyond the boundary of the designated site. Hence the importance of considering these as functionality linked to the designated site areas and ensuring adequate consideration is made of them when assessing potential impacts.

3.118 Vegetated shingle is listed as a priority habitat under the UK Biodiversity Action Plan (JNCC 1994)[73]. The Vegetated Shingle Habitat Action Plan recognises two principle issues (2.1 and 2.2 below) relating to sea defence and the sustainability of shingle habitats:

> *"2.1 Sediment supply. The health and ongoing development of a shingle feature depend on a continuing supply of shingle. This may occur sporadically as a response to storm events rather than continuously. It is frequently lacking owing to interruption of coastal processes by coast defence structures or by artificial redistribution of material within the site (e.g. Dungeness). Attempts have been made to rectify the situation by mechanical reprofiling, which is likely to fail in the long run because it does not address the lack of new material, or by beach recharge."*

3.119 Studies[74] tend to confirm the view that in many instances engineering solutions to shingle erosion are unsustainable in the medium to long term, leaving the consideration of approaches which 'work with nature' and accept the 'naturally' dynamic state of shingle beaches.

> *"2.2 Natural mobility. Shingle features are rarely stable in the long term. Many structures exhibit continuous longshore drift and ridges lying parallel to the shoreline tend to be rolled*

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[72] Doody, Dr P. & Randall, Dr R. (2003) *A Guide to the Management and Restoration of Coastal Vegetated Shingle.* English Nature report

[73] JNCC (1994) Biodiversity. The UK Action Plan. HMSO. London

[74] Specifically Doody and Randall (2003)

over towards the land by wave action in storm events. This movement has a knock-on effect on low-lying habitats behind the shingle. Movement is likely to be accelerated by climate change resulting in sea level rise and increased storminess."

3.120 The reliance on a 'static' approach to conservation, which is so often associated with the identification of boundaries of sites on maps, may be counter-productive for a habitat existing in such a dynamic environment and the importance of functionally linked areas as mentioned above. Removing static artificial features such as groynes and breakwaters and/or ceasing beach nourishment could help bring about major restoration of shingle beaches.

3.121 The UK Biodiversity Action Plan's Vegetated Shingle Habitat Action Plan Target 1 is –

> "Prevent further net loss of existing vegetated shingle structures totalling about 5800 ha. (However local gains and losses due to storm events occur sporadically and should be accepted provided that the national and regional resources are maintained overall.)

3.122 Doody and Randall 2003[75] define a primary concern must be the conservation of the remaining areas of relatively intact and undisturbed areas of vegetated shingle in the UK. All damaging activities must be prohibited on all sites where important areas of this extremely rare habitat exist. The UK has a particularly important role to play here by virtue of the extent of the shingle (gravel) banks that occurs and their world significance.

3.123 Any acceleration of erosion that could lead to a breach in the shingle and dune barrier bank, allowing saline inundation of the low-lying marshes inland would significantly affect the freshwater interest features within RSPB Minsmere. This would extend to SPA designated wetland breeding birds and SSSI bird, flora and invertebrate communities.

Summary of concerns relating to the assessment of coastal processes

3.124 Use of the Rochdale envelope approach has been deployed by the Applicant and said to explain why detailed designs on the coastal defences were not available when the Application was submitted. In the absence of these designs we have been unable to be confident that the approaches to the coastal defences and the interaction with the neighbouring SAC habitat is robust and would conclude that the precautionary principle needs to be followed. This is further emphasised by the inherent uncertainty when predicting likely coastal geomorphological outcomes in this dynamic environment.

3.125 In addition, the RSPB does not agree that the assessments are sufficiently robust. The Applicant have agreed to undertake 'additional bespoke modelling' (see *Written summaries of SZC Co.'s oral submissions at Preliminary Meeting Part 1*[76]) but it is our view that this should not be determined to be 'additional' or 'bespoke' as we believe that the work is necessary to provide a satisfactory environmental assessment given the risk of potential impacts to the Minsmere – Walberswick designated sites.

3.126 It is our view that the potential impacts arising from the Application, should be considered in combination with natural variation. We believe this is the appropriate precautionary approach and should be adopted.

3.127 We recognise that the Applicant is preparing updated models in respect of the revisions to the HCDF, SCDF and BLFs, and in the absence of these we do not believe there has been adequate

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[75] Doody, Dr P. & Randall, Dr R. (2003) *A Guide to the Management and Restoration of Coastal Vegetated Shingle.* English Nature report

[76] *Written summaries of SZC Co.'s oral submissions at Preliminary Meeting Part 1 (PDB-012)*

assessment on the potential impacts on the Minsmere – Walberswick designated sites through both the ES and HRA processes, as a consequence of potential impacts being screened out. We believe that the current lack of confidence in the modelling and assessment should require a precautionary approach so that the true worst-case scenario is considered for the designated sites and appropriate mitigation measures considered.

3.128 *BEEMS Technical Report TR403 and the Volume 2, Chapter 20 of the ES [APP-311]* identified that, if no further intervention was undertaken, shoreline recession was likely to erode the soft coastal defence feature (SCDF) and expose the hard coastal defence feature (HCDF) within the timeframe of 2053 – 2087 (i.e., within the Sizewell C Project's operational phase). An exposed HCDF, dividing the otherwise continuous shingle beach in two, would partially or fully block the longshore shingle transport corridor. If persistent, shingle starvation and erosion either side of the exposed HCDF would be expected.

3.129 Experience along the Suffolk Coast demonstrates that 'hard' features on the coast influence the shoreline to the north and south. With the HCDF anticipated to become a marine feature at some point in the operational life of SZC it is very likely to affect the coastal geomorphology of the Minsmere frontage. Therefore, we cannot agree with the Applicant's conclusion that no possible impacts are predicted.

3.130 We understand that the purpose of soft coastal defence feature (SCDF) seaward of the HCDF is to mitigate any impacts on sediment transport or coastal processes by maintaining a continued source of accretion during major storm events and that no coastal geomorphological impacts are predicted along the Minsmere frontage with this mitigation in place.This is confirmed in the Applicant's submission (e.g. Volume 3 Chapter 2 Environmental Statement Addendum Coastal Geomorphology and Hydrodynamics Appendix 2.15.A section 6.1)

3.131 The lack of sufficient detail currently on the management of the SCDF provides us with limited confidence in the conclusion that the HCDF will remain a terrestrial feature for much of the operational life of SZC and that the potential impacts can be satisfactorily mitigated.

3.132 We await details of the revised modelling to determine whether the aspects such as the potential value of southeast waves in aiding accretion to the Minsmere shoreline have been considered.

3.133 The suggestion that the southern Minsmere frontage no longer supports the designated Annual vegetation of drift lines and Perennial vegetation of stony banks due to the loss of SSSI Site Unit 113[77] (which has been destroyed through coastal erosion) is incorrect. The implication that Natural England have concluded that any designated vegetation features have been lost from this stretch of the coast is misleading. Natural England's site unit assessment for Unit 113[78], provides a more accurate description of the situation, concluding that the unit has been destroyed through natural processes[79]. It suggests that Unit 113 may have rolled back into Unit 112[80] (which is what you would expect to have happened). It describes Unit 112 as:

> *'Annual shingle vegetation was evidence but appeared to be a single species of Atriplex (Atriplex prostrata). The target should be reconsidered as other shingle sites have shown a*

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77 *Volume 2 Main Development Site Chapter 20 Coastal Geomorphology and Hydrodynamics Appendix 20A Coastal Geomorphology and Hydrodynamics: Synthesis for Environmental Impact Assessment* (APP-312) *Section 1.1 p15*

78 Natural England's site unit assessment for Unit 113

79 "This unit has been lost through coastal erosion…site visit confirmed that all the unit had gone or if deposition had started to occur again no shingle habitat was present…It may be that Unit 113 has rolled back into Unit 112. A sea wall divides Minsmere Scrapes (Unit 48) from the beach (Unit 112) and supports dune communities. It is likely that if the shingle is trying to rollback back this is inhibitated by the seawall and therefore coastal squeeze could be an issue for unit 112."

80 Natural England's site unit assessment for Unit 112

low diversity of species (notably Orfordness, in discussions with Sue Rees). Perennial shingle vegetation was present including Rumex crispus, Crambe maritime (sic) and Glaucium flavum, all abundant or frequent. Bitter stonecrop and sea sandwort present.'

3.134 Emma Hay, the Natural England officer, who conducted the site assessment has confirmed (*pers comms*) that the 'loss' of Unit 113 should be interpreted as a geographical loss of this unit and it should not be considered that the SAC vegetation interest feature has been lost from this frontage[81].

3.135 This is further confirmed by Cadbury (199682, 200483 & 201584), alongside the NE site assessments which confirm that the southern Minsmere frontage between the sluice and the red line development boundary support the two important vegetated shingle communities: summer annuals (*Atriplex* spp.) reflective of a beach stable over the growing season only and short-lived perennials reflecting a beach stable for 3+ years. Doody and Randall (2003)[85] confirm that vegetation is primarily controlled by the proportion and size of the fine fraction material of the matrix under 2mm diameter, which is critical for vegetation germination and seedling stages. The species characteristic of the Minsmere frontage are those associated with no matrix (e.g. *Atriplex glabriuscula, Crambe maritima, Lathyrus japonicus*) or a sand influence (e.g. *Honckenya peploides*).

3.136 We note that the likelihood of the development accelerating a breach in the barrier bank is considered to be unlikely. However, although this eventuality has low likelihood, it would be a significant risk to the freshwater habitats were it to occur.

Comments on implications of the changes to the application on coastal processes

3.137 The fact that the design parameters have changed significantly between the original Application and the Change Application does cause us concern about the robustness of the assessments including ensuring that the worst-case scenario has been considered. Whilst we appreciate that Addendums for both the ES and HRAs have been provided (and we appreciate the Applicant's signposting document[86] to help navigate between the many volumes and appendices) there do still appear to be some gaps. For example, the movement of the toe of the HCDF 10m seaward of the Application's design at the abutment to the enhanced permanent BLF[87], along with the introduction of a permanent BLF and changes to the temporary BLF require a modelling and further assessment. We do not agree that the application of expert judgements, as proposed in the Environmental Statement Addendum [88] are sufficient.

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81 Unit 112 "Minsmere Haven South is a shingle beach backed by sand dune vegetation, comprising some mobile dune community of marram grass and fixed dune grassland behind. Shingle vegetation is sparse within the unit due to coastal process/erosion of the beach with annual species limited to occasional spear-leaved orache and some sea sandwort, plus occasional perennial vegetation species - sea kale, sea poppy, sea holly and sea pea….. The Unit was found to be in favourable condition and passed on all its targets for Mobile & Embryonic Dune Feature & Fixed Dune Grassland features…"

82 Cadbury, C. J. (1996) Vegetation survey of the Coastal Grazing Marshes and Dunes at Minsmere RSPB Nature Reserve, 1993. *Unpubl. rept. Reserves Ecology.* RSPB, Sandy

83 Cadbury, C. J. (2004) Repeat vegetation survey of the Beach and Dunes at Minsmere RSPB Nature Reserve, Suffolk, 2004. *Unpubl. rept. Reserves Ecology.* RSPB, Sandy

84 Cadbury, C.J. (2015) Assessing the Impact of the Tidal Surge of 5/6 December 2013 on the Shore Profile and Flora of RSPB Nature Reserves on the Norfolk and Suffolk coast. *Unpubl. rept. Reserves Ecology.* RSPB, Sandy

85 Doody, Dr P. & Randall, Dr R. (2003) *A Guide to the Management and Restoration of Coastal Vegetated Shingle.* English Nature report

86 The Applicant's helpful Navigation Document - REP1-002

87 Environmental Statement Addendum Volume 1: Environmental Statement Addendum Chapters, Chapter 2 Main Development Site 2.15.18 p240 PINS ref AS-181

88 2.15.9 page 237 and 2.15.15 page 238

3.138 We are aware that the Applicant is in the process of some additional modelling of the potential impacts of the proposed changed structures and operation. The changes in the coastal and nearshore marine environment provide no improvement to the potential threats identified above and in our view, increase the potential risk of impacts. This is a consequence of:

- Confirmed designs for the HCDF and SCDF still awaited
- Seaward movement of the proposed toe of the HCDF
- Earlier intervention required to manage the SCDF[89]
- Increased length of the permanent BLF and design changes to enable operation
- Introduction of an additional temporary BLF
- Increase in vessel deliveries and therefore potential impact of vessels on coastal processes

3.139 The change document has relocated the HCDF seaward of the Application site, which has increased these concerns as it would be assumed that the HCDF will become exposed ahead of the timeframe quoted in the Application.

Conclusions regarding impacts on coastal processes on protected sites and species

3.140 In conclusion, based on the above concerns and the evidence provided at this stage, we do not agree that *adverse effect on integrity* can be ruled out for the following sites and features for impacts from the project alone:

- Annual vegetation of drift lines (Minsmere to Walberswick Heaths & Marshes SAC);
- Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC);
- Little Tern (Minsmere-Walberswick SPA);
- Breeding waterbirds (Shoveler, Gadwall, Teal, Bittern, Marsh Harrier, Avocet, Bearded Tit) (Minsmere-Walberswick SPA)

Mitigation and monitoring

Lack of detailed design

3.141 In the absence of a confirmed design approach and full assessment of all *likely significant effects*, it is difficult to evaluate whether the Coastal Processes Monitoring and Mitigation Plan (CPMMP - *Volume 3 Chapter 2 Environmental Statement Addendum Coastal Geomorphology and Hydrodynamics Appendix 2.15.A*) will be appropriate in scope.

3.142 We have immediate concerns regarding the approach to embedded mitigation using the SCDF through the CPMMP. We consider that the design detail of the SCDF needs to be confirmed before the end of the Examination.

Mitigation activity in response to triggers

3.143 In addition, due to the dynamic coastal environment, the CPMMP must be developed in a manner that allows impacts due to the Application to be appropriately addressed. As mentioned above we do not agree, with the assessment of minor or insignificant impacts and therefore have additional concerns as to how trigger points for intervention may be defined and considered within the CPMMP.

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[89] Earlier intervention to maintain the SCDF has been proposed as a consequence of the seaward movement of the HCDF and therefore the anticipated earlier erosion of the soft coastal defence

3.144 We welcome the commitment[90] that the nearshore bars will be monitored to assess any impacts of the FRR and CDO beyond the predictions in the models, but note that the CPMMP does not appear to identify any mitigation strategy for these nearshore bars, should impacts be identified. Due to the potential to increase the vulnerability of the shoreline if unexpected changes to the nearshore bars occur, it is vital that measures are proposed now.

3.145 In addition, we think it is necessary to consider some form of adaptive mitigation for example the replenishment of the nearshore bars (and therefore maintain protection to the shoreline) should the monitoring reveal predicted impacts.

Mitigation activity and designated sites

3.146 In the CPMMP, section 6.2 Beach management framework, the monitoring framework in Table 1 indicates that monitoring will take place north up to the Minsmere outfall. We welcome the commitment to monitoring the southern RSPB Minsmere frontage, but note that no attempt has been made to understand how mitigation would be implemented if this monitoring established an unexpected adverse impact on the Minsmere frontage. We note also that the CPMMP commits to providing final proposals for monitoring the annual vegetation of drift lines but are unaware of any further details being presented.

3.147 Section 6.5.4 (page 36) of the CPMMP states that

> *no sediment would not be extracted from statutory designated sites (in the cases of bypassing or beach recycling) unless accumulating sediments were a direct effect of the Sizewell C Project (mitigation or presence of the HCDF) and approval was given following demonstration that designated features would not be affected*.

3.148 Sediment accumulation as a direct effect of Sizewell C would represent an *adverse effect* and we are yet to understand how this could be addressed without impacting on designated features as described above.

3.149 Section 6.5.4 also states

> *sediment would not be deposited on the supra-tidal beach within statutory designated sites unless approval was given following demonstration that designated features would not be affected*.

3.150 Again, if this requirement proved necessary, due to effects from the Application exceeding the model predictions or via a need to maintain the SCDF, we do not understand how it could be implemented without affecting the designated features and so do not believe we have enough evidence to support this approach.

3.151 In addition, the approach described in the CPMMP recognises there is the potential for impacts on the RSPB Minsmere frontage north of the red line boundary. Should the CPMMP identify the need for interventions in this area, the Applicant currently (as far as we are aware) has no powers to implement such required measures. As well as our ecological concerns as a neighbouring landowner (as discussed below) we are keen to know now whether land owned by the RSPB could be required.

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[90] Section 4 p 22-23 CPMMP AS-237 6.14 ES Addendum Volume 3: Addendum Appendices Chapter 2 Main Development Site Appendix 2.15.A Coastal Geomorphology and Hydrodynamics

3.152 We also welcome the commitment to monitor the annual vegetation of drift lines habitat recognising their dependence on coastal geomorphology (i.e., supra-tidal shingle)[91]. However, we are concerned that this dependence has not been captured in the HRA and therefore again concerned about how any trigger points for intervention will be determined.

3.153 In addition, it is important to know now how any evidence of impacts identified by the monitoring would be mitigated. We are not familiar with successful mitigation techniques being implemented in coastal shingle habitats to enhance the annual vegetation of drift lines, where changes in coastal geomorphology are creating adverse effects and therefore have concerns about what might be viable and effective mitigation.

3.154 The statement in section 6.1.1.2 of the CPMMP that the *activities/pressures associated with the use of the SCDF include heavy plant on the beach* also requires greater clarification. At present the red line boundary for the Application abuts the southern boundary of the Minsmere-Walberswick SPA, SAC, Ramsar and SSSI. Without more detailed information regarding the activity of this heavy plant in this sensitive location immediately adjacent to the designated sites we cannot be confident in the conclusion that this activity does not have the potential to impact the designated features.

3.155 What mitigation is proposed[92] appears to be solely focussed on the maintenance of the shingle transport corridor, with the main aim of providing protection for the HCDF. It is unclear how this proposed mitigation for sea defence purposes will be aligned to avoiding *adverse effects* on the designated flora or the integrity of the neighbouring frontage to the north, especially as impacts on the frontage could have significant effects on the integrity of the Minsmere – Walberswick designated sites.

3.156 As mentioned above, it appears that because ES Volume 2, Chapter 20 of the ES [APP-311] showed no significant effects on coastal geomorphology receptors from the SCDF, the potential for effects of this activity on the designated sites' features, such as annual vegetation of drift lines, may not have been adequately taken into consideration. This is of increased concern given the acknowledged value of the existing frontage in front of the proposed SZC site as set out in section 7 of the CPMMP.

Overall approach to Monitoring and Mitigation

3.157 The final paragraph in section 6.1.1. of the CPMMP states:

> *In prolonging the period before any mitigation is needed, a longer evidence base on aspects such as storm erosion and recovery under rising sea levels will be developed, which in turn will lead to more effective mitigation decisions. During this period there may also be advances in our understanding of mixed sandgravel beaches like Sizewell, and in the models used to predict how they respond in the short and long term, which could be used where appropriate to make further refinements to the final monitoring plan.*

3.158 Whilst these aspirations are laudable, we believe this level of uncertainty is unsatisfactory and does not provide confidence in the process. The Applicant and their predecessors have been monitoring shoreline evolution and processes on the Sizewell frontage for more than two decades through the auspices of the Sizewell Shoreline Management Planning Group. Given that after two decades of study this appears not to have provided a sufficient evidence base for making

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[91] Section 7 page 38 CPMMP - AS-237 6.14 ES Addendum Volume 3: Environmental Statement Addendum Appendices Chapter 2 Main Development Site Appendix 2.15.A Coastal Geomorphology and Hydrodynamics

[92] Section 6.5 p 33-36 CPMMP AS-237 6.14 ES Addendum Volume 3: ES Addendum Appendices Chapter 2 Main Development Site Appendix 2.15.A Coastal Geomorphology and Hydrodynamics

confident predictions regarding potential impacts and appropriate mitigation, we remain concerned that this aspiration is achievable.

3.159 The statement in section 6.1.1.2 of the CPMMP:

> *During construction of Sizewell C, the sedimentary material presently in this location* (the SCDF) *will be impacted as required for landscaping, building the SCDF and some aspects of the HCDF or BLF (e.g., the installation of the BLF decking at c. 5.2 m ODN across the 5 m bund and/or SCDF which has the same elevation).*

3.160 We request more detail on this statement. In combination with the statement in section 6.1.1.1

> *Coffer dams may be required to place the initial toe of the HCDF, but they would be set further back and are unlikely to be exposed to the sea while the HCDF is built, so long as the present beach width and elevation are maintained*

we are not reassured that there is an explicit requirement to retain the shingle feature at this location during construction. Given that the shingle/beach feature in this area has the potential to support annual vegetation of drift lines (as highlighted in section 7 of the CPMMP) and is also integral to the access and amenity planning/recreational displacement in terms of public access during the construction period, we seek a stronger commitment that the feature will be maintained. As noted above, Doody & Randall (2003)[93] highlight the removal of gravel from the beach itself as the most damaging activity affecting coastal vegetated shingle.

3.161 In addition, the CPMMP does not appear to have been revised to address issues arising from the revised approach to coastal defences and infrastructure arising from the Change Application.

Hydrology (surface water impacts)

Introduction

Minsmere RSPB Reserve

3.162 The hydrological management of Minsmere RSPB Reserve is ultimately reliant on the Minsmere Sluice, located at the seaward end of the Minsmere New Cut. The Minsmere Sluice drains the catchment of the Sizewell Marshes SSSI and the Minsmere Level.

3.163 The importance and features of the Sizewell SSSI are described at above but it is worth noting here the importance of water levels for this SSSI and its features (as discussed further below). For example Natural England's list of *operations likely to damage* relevant for this SSSI[94] include:

> 13b: Modification of the structure of watercourses (e.g. ditches, dykes, drains), including their banks and beds, by re-alignment, re-grading and dredging.
>
> 14: The changing of water levels and tables and water utilisation including irrigation, storage and abstraction from existing water bodies and through boreholes.
>
> 15: Infilling of ditches, dykes, drains or marshes.

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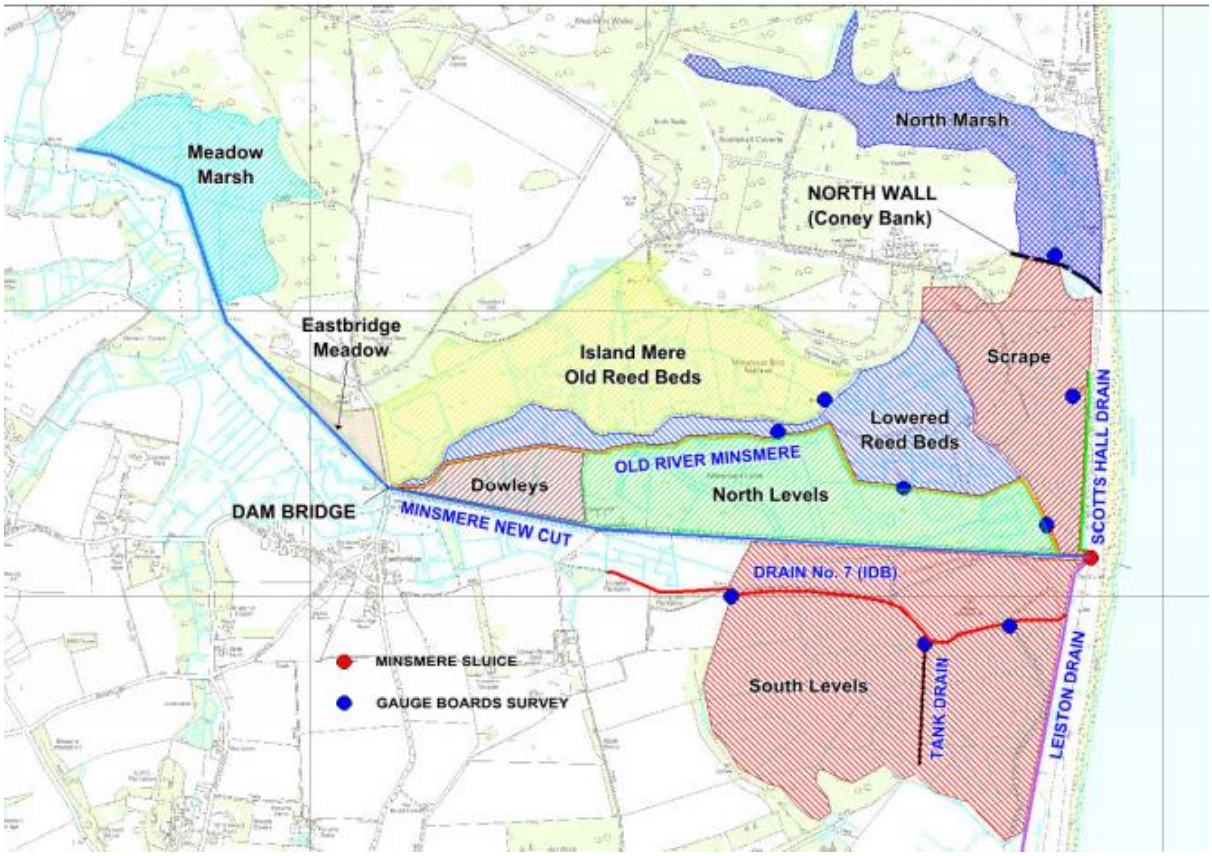
93 Doody, Dr P. & Randall, Dr R. (2003) *A Guide to the Management and Restoration of Coastal Vegetated Shingle.* English Nature report

94 Sizewell Marshes OLD - https://designatedsites.naturalengland.org.uk/PDFsForWeb/Consent/1003416.pdf

3.164 The surface water on the marshes south of the New Cut (South Levels) are controlled via the Internal Drainage Board's (IDB)[95] Drain No 7 which is connected to the Leiston Main Drain which discharges into the southern chamber of the Minsmere Sluice.

3.165 The surface water on the land north of the New Cut (North Levels) drains into the Old Minsmere River which drains into the New Cut.

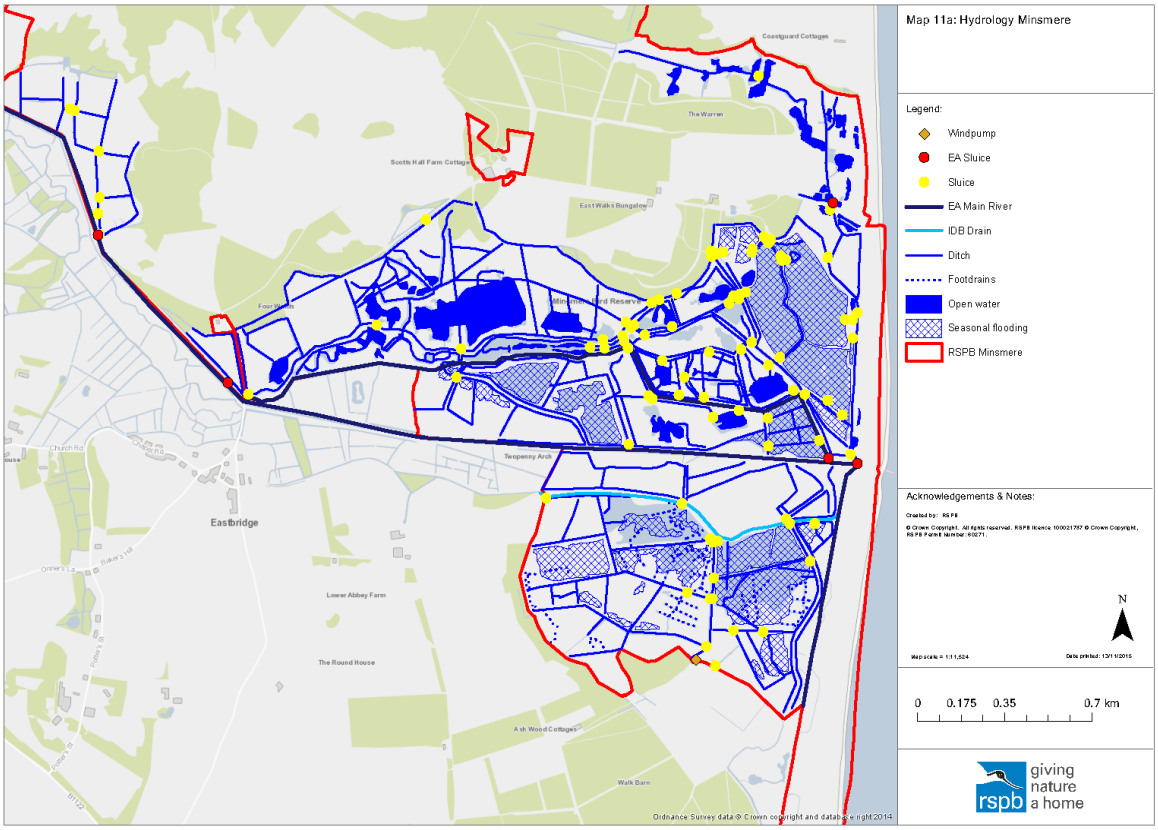
3.166 The water levels in the main reedbed, Scrape and north marsh drain into the Scott's Hall Drain which is discharged via a syphon into the southern chamber of the Minsmere Sluice. The New Cut is discharged via the northern chamber of the Minsmere Sluice. The New Cut can also be discharged via the southern chamber of the sluice during periods of high flow.



Map 1: Locations of key drains and water level monitoring gauge boards (from Main Development Site Flood Risk Assessment Appendices 1- 7 Part 1[96])

[95] Ultimately the IDB is are responsible for the maintenance of the main channel that drains water from the South Levels, but please note that the RSPB have control to manage water levels on the South Levels 'upstream' of this drainage ditch

[96] From p212 of Main Development Site Flood Risk Assessment Appendices 1-7 Part 1 (APP-094)



Map 2: Locations of water control structures

3.167 Any increase in water flows or volume in the Leiston Main Drain has the ability to affect the management of water levels on the RSPB Minsmere South Levels by impeding the rate of discharge from IDB Drain No 7, potentially causing surface water to 'back up' on the South Levels marshes. An increase in flows or volume in the Leiston Main Drain could also reduce the rate of discharge in the Scott's Hall Drain, thus holding surface water levels higher in the main reedbed, Scrape and north marsh on RSPB Minsmere.

The relevance of water levels for the Ecology of RSPB Minsmere[97]

3.168 The primary inputs for the Minsmere Levels are from a predominantly arable catchment, with some heath and mixed woodland. Direct run-off from adjacent arable land is mostly intercepted and diverted by the western boundary ditch that forms the western and southern boundary to the South Levels. Rainfall also provides a significant input and, in extreme events, floodwater from the New Cut and Old Minsmere River (EA Main Rivers) also provide surface water. There is a saline influence to the east. This may be from groundwater beneath the dunes, occasional leaks through the Minsmere Sluice and possibly sea spray.

3.169 The main input to the **reedbeds** are via groundwater springs and rainfall. Water levels within the reedbed are maintained at an optimal depth of 20-30 cm depth throughout the late winter and early spring. Levels drain down as a consequence of evapo-transpiration during most summers. Levels are lowered between September and December when reed-cutting management is taking place. The reedbed water level management is principally to benefit key breeding bird species such as bittern.

97 The areas, features in bold are included in the maps above

3.170 The main inputs to the **coastal lagoons** are from rainfall, freshwater from North Marsh and the eastern reedbed and brackish water from the sea via the Minsmere Sluice (an EA structure). The coastal lagoons are allowed to draw down slowly over late spring and summer, to provide suitable conditions for breeding avocet. They are lowered in late summer to allow management of vegetation on islands, and are maintained at varying depths in autumn and winter to provide a range of feeding and loafing conditions for waders and wildfowl.

3.171 The main inputs to the **South Levels** are from rainfall and groundwater springs. The grazing marsh areas are flooded for winter wildfowl. Other areas remain damp and, if possible, unflooded. Water levels are drawn down through the spring and summer for breeding waders and lowered to allow grazing and mechanical operations as the summer progresses.

3.172 The **wetlands north of the New Cut** drain east and south. The **North Levels** drain into the Minsmere Old River (an EA Main River) and then into the New Cut via an EA sluice. **The Scrape**, **North Marsh and the eastern reedbed** areas drain through the Minsmere Sluice via a chamber separate to the New Cut. Either route can drain the **western reedbed areas**. During periods of heavy rainfall, an EA penstock is operated to give priority to draining the New Cut and this can prevent drainage from the Scrape, North Marsh and eastern reedbed. Heavy rainfall can also cause the New Cut to overtop its banks, displacing water into the western end of the reedbed and Old Minsmere River.

3.173 The **wetlands south of the New Cut** drain east and north. They drain into IDB Drain No. 7 and the Leiston Drain (an EA Main Drain), then through the Minsmere Sluice via the same chamber as the Scrape. **Eastbridge Meadow and Meadow Marsh** drain directly into the New Cut via EA sluices. During periods of heavy rainfall, the New Cut banks near the sluices as well can be over-topped, with water flowing into Eastbridge Meadow and Meadow Marsh.

3.174 Water levels are controlled by structures at the locations shown in Map 2 (above). Most of these are piped sluices in earth banks.

3.175 Levels are monitored at gauge boards shown on Map 1 (above). Salinities are monitored at seven locations in the reedbed, and each of the three main coastal lagoons

3.176 Since 2008, following episodes of heavy rain, the Minsmere Sluice, which is tidally operated, has proved unable to evacuate the consequent run-off quickly enough to prevent detrimental flooding of the reserve's wetland habitats and some visitor trails, as happened in May/June 2012 and in several recent winter periods. The Environment Agency have undertaken improvement works to the main sluice during 2012 – 2014, but these will not always prevent these periodic flooding occurrences. Given this known risk, it is imperative that additional increases in flow are not introduced into the system that may further compromise the ability to manage target water levels following extreme events.

3.177 The risks of increased water levels on the designated bird species of the Minsmere to Walberswick SPA includes reduced breeding success, particularly if there are prolonged periods of raised water levels in April - June. Species particularly prone to nest loss through flooding events are bittern, avocet, common, Sandwich and little terns. Spring flooding has caused nesting failure in several springs in the early 2000s and it is imperative that additional factors that could increase the risk of this situation are not introduced by the Application. Nest failure is caused when the eggs are submerged by flood water for a protracted period. Whilst eggs may tolerate short periods of inundation (such as being flooded by a high tide event) they are not able to survive being flooded for a period of many hours. Equally the inability of the adults to incubate the eggs when they are submerged and of course the very serious issue of nests and eggs being washed away by flood

water. An increase in nest failure events can obviously have a detrimental impact on the species population.

3.178 Foraging opportunities for some Minsmere to Walberswick SPA and Ramsar site designated bird species can also be compromised by changes in water level. Bearded tit (Ramsar feature) population survival has been demonstrated to be adversely affected by raised water levels in the non-breeding season preventing access to food (Wilson & Peach 2006)[98].

3.179 Loss of nest sites for example for avocet and terns who nest on bare substrates exposed by receding water levels, which are at threat of being submerged beneath raised water levels. Also raised water levels can lead to the delaying of breeding attempts. Again, this can result in reduced breeding productivity if the birds only have a shorter breeding season in which to attempt breeding, which, if repeated, could have an effect on local populations over time.

Concerns regarding potential impacts on water levels of the Minsmere to Walberswick designated sites

3.180 The potential for the Application to alter the surface water flow regime and lead to impacts on the Sizewell Marshes SSSI and Minsmere to Walberswick Heaths and Marshes SSSI, SAC, SPA and Ramsar site is acknowledged in table 19.9 on p52 of ES *Vol. 2 Ch. 19 Groundwater and Surface Water*[99].

3.181 Table 5.2 item 3b (p79) of *Shadow HRA Vol. 1: Screening and Appropriate Assessment Part 1*[100] provides a justification for the decision that likely significant effects cannot be excluded, confirming that changes to direct hydrological connections, water levels and water quality/chemistry could be involved for example

> '*A number of factors mean that the potential for measurable hydrological impact as a result of construction (or decommissioning) activities on either side of the sluice is minimal but any increase in water levels at the Minsmere Sluice could potentially affect surface water levels to the north. As such, discharges into watercourses within the development area, or changes in water levels, flow, ditch alignment or sedimentation rates within the watercourses, could have a direct effect on water quality in the Minsmere South or Minsmere North Levels.*

and

> *interrupt*[ions] *or* alter[ations to] *the baseline groundwater regime…could potentially change the hydrological or hydrogeological properties of the site and adjacent land and, in turn, the conditions that support wetland habitat….The potential for direct effects on water quality…. means that likely significant effects cannot be excluded for those qualifying features dependent on wetland habitats.'*

3.182 The operation of the Minsmere Sluice means that the potential impact to the north of the New Cut would be greater than just affecting the Minsmere North Levels. This is because any impediment on the drainage of the Scott's Hall Drain at the Sluice could impact on water levels in the main reedbed, Scrape and north marsh.

3.183 Minsmere Sluice allows some entry of seawater into Minsmere to help maintain the Ramsar designated brackish transition habitats in the ditches and appropriate salinities on the scrape. This means that these habitats are vulnerable to any contaminants entering by this route. The

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98 Wilson, J. & Peach, W. 2006. Impact of an exceptional winter flood on the population dynamics of bearded tits (*Panurus biarmicus*). Animal Conservation 9:463-473

99 ES Volume 2 Main Development Site Chapter 19 Groundwater and Surface Water (APP-297)

100 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 (APP-145)

Application would discharge hydrazine, which is used as an anti-foulant in the cooling water system, during the later stages of construction and throughout its operation. Paragraph 21.6.72 of ES Ch. 21 Marine Water Quality and Sediments (AS-034) explains that at certain states of the tide it is possible that low levels of hydrazine would enter Minsmere via the Sluice. Given the toxic properties of hydrazine, we consider that this requires further consideration through the HRA. This issue is discussed in more detail in our comments on marine ecology below.

3.184 We are grateful to the Applicant for the detailed and technical discussions concerning our hydrological concerns and believe it will be submitting further information in due course. For now therefore we have set out our concerns but will of course review any new information or material and provide further comments.

3.185 All these potential effects could impact the breeding attempts of species such as avocet and bittern (SPA, Ramsar and SSSI species) by flooding nests or affecting feeding sites.

Proposed Primary Mitigation to manage potential impacts

3.186 Section 7.8.15 – 19 (pp294-295) *Shadow HRA Vol. 1: Screening and Appropriate Assessment Part 1*[101] states that due to the potential risk of greater volumes needing to be discharged down Leiston Drain to maintain water levels within the Sizewell Marshes SSSI potential inundation of habitats within the Ramsar site could occur and prevent the control of water levels. Adding that:

> *If the increased discharge flows are sufficient to reduce available capacity in the southern chamber of the Minsmere Sluice, this could cause back flooding within the Scott's Hall Drain, potentially affecting the Minsmere to Walberswick Heaths and Marshes SSSI, part of which is included within the Ramsar site.*

3.187 Therefore the Applicant has included mitigation measures within the surface water management design (during the construction period), arguing that these along with the proposed Sizewell Drain realignment isolate the proposed development from the surrounding areas (Section 7.8.15 – 19 (pp294-295) adding:

> *The mitigation measures also ensure that any flows discharged to an existing surface water receptor would have passed through water quality treatment measures and would be discharged at greenfield rates.*[102]

3.188 Within the same section (Section 7.8.15 – 19 (pp294-295) the Applicant states:

> *The proposed water management structures would also allow for easy manipulation of the water levels and flows and thus levels/flows within the Leiston Drain could be reduced as and when required to allow for the Scott's Hall Drain to discharge efficiently when required. This coupled with the fact that the relatively small contribution from Leiston Drain to the overall flow at Minsmere Sluice, limited effects on the flow regime of the southern chamber of the Minsmere Sluice are anticipated and thus no significant effect is predicted for the Scott's Hall Drain and associated drainage network. It is, therefore, anticipated that the proposed development should create no significant effect on the flow regime or water quality of the existing surface water receptors and, therefore, no effect on the habitats within the Ramsar site.*

3.189 Repeatedly concluding in section 8 (e.g. page 393, section 8.8.4) that with mitigation in place to manage surface water discharges, as described in section 7.8.b (*sic*) of this report above, there

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101 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 (APP-145)

102 i.e. the flow rates of water discharging from the construction surface will be managed to not exceed the rate that would occur in natural circumstances

would be *no adverse effects on the integrity* on any of the Ramsar qualifying habitats within the Minsmere-Walberswick SPA.

3.190 And since these habitats also support the SPA qualifying bird species, it is concluded that there would not be an *adverse effect on the integrity* of any of the SPA qualifying feature either, including the breeding avocet, during the construction or decommissioning phases.

3.191 We cannot see how a situation of raised water levels throughout the area, particularly in extreme events, can be managed by the proposed mitigation to avoid impacts successfully. In such events, it is inevitable that the mitigation structure controlling the levels in Sizewell Marshes SSSI will need to be adjusted to allow increased discharge. In such circumstances we question how the Applicant proposes to reduce levels/flows in the Leiston Drain as described above to avoid increasing flows into the Minsmere Sluice and affecting water level control at RSPB Minsmere. In extreme events this scenario increases the risk of flooding the nests or reducing feeding opportunities for SPA, Ramsar and SSSI species such as avocet and bittern.

3.192 *Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology*[103] repeats on page 23 that the realignment of the Sizewell Drain and associated water control features would allow for control of the Sizewell Marshes SSSI water levels[104]. Adding that

> *Control structures would include passage for eels and other fish (see Eels Regulations Compliance Assessment, 2019).'*

and on page 24

> '*Construction infrastructure would be in place to ensure all surface runoff and foul water is captured and treated and does not enter adjacent designated sites. Ditches, bunds and swales would be constructed to prevent untreated surface water run-off from leaving the site. Oil/petrol interceptors would be incorporated into the drainage design. Where complete infiltration to ground is not feasible, Water Management Zones (WMZs) have been embedded into the design. These systems would be designed to discharge treated water to the surface water drainage network at greenfield run-off rates. Foul water would be pumped to a central treatment plant, prior to discharge to sea. Run-off would be managed as part of the Construction Drainage Strategy, which would include mitigation measures to manage surface water discharges*[105]

3.193 We understand that further information is to be provided to the Examination and will comment further on this once we have had time to review.

3.194 Section 19.5.1 (page 41) of *ES Vol. 2 Ch. 19 Groundwater and Surface Water* [106] does state

> '*primary mitigation measures have been identified through the iterative EIA process and have been incorporated into the design and construction planning of the proposed development. Tertiary mitigation measures are legal requirements or are standard practices that would be implemented as part of the proposed development.*'

3.195 However we question this assertion since *detailed* mitigation measures to manage surface water flows have yet to be progressed in detail and therefore based on the information currently before

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[103] Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033)

[104] As set out in more detail in *Chapter 19 Ground and Surface Water of the ES (Doc Ref. Book 6)*

[105] With *Chapter 19: Groundwater and Surface Water* and *Chapter 22 Marine Ecology and Fisheries* setting out the full details.

[106] ES Volume 2 Main Development Site Chapter 19 Groundwater and Surface Water (APP-297)

the Examination could not be described as *incorporated in the design…. of the proposed development*. We have detailed these concerns further in section 4.3.4 below.

3.196 There are some further details for example in Section 7.4 (pages 50-51) of ES *Vol. 2 Ch. 19 Appendix 19B*[107]we find the following:

> '*Where it is anticipated that groundwater levels would rise to an unacceptable degree as a result of the development…engineered mitigation features, such as passive drain systems, and/or ditch maintenance will be considered to control groundwater levels.*

3.197 But as far as we are aware there is currently no firm commitment to these measures but for this reference in the Environmental Statement.

3.198 There is confirmation that monitoring of surface water and groundwater and the efficacy of the mitigation measures as well and further assessment will be carried out for example to determine the extent of the predicted change and to allow further mitigation *to be designed where necessary*.

3.199 Adding that

> *The monitoring network will provide the capability for early identification of potentially unacceptable changes in flows and levels allowing control measures to be implemented.'*

3.200 However, to be able to rely on such commitments and the effectiveness of any additional/secondary mitigation proposed – full details must be provided now and included within DCO conditions.

3.201 Similarly, section 19C.4 *Water level control structures* (pages 4-5) of *Volume 2 Main Development Site Chapter 19 Appendices 19F* [108] describes the need for water control structures and suggests tilting weir[109] or piped structure[110] for the realigned Sizewell Drain. We welcome recognition that there is a need for water level control structures to manage the Sizewell Marshes SSSI.

3.202 However, the described approach is limited to the raising or lowering of water levels to manage potential adverse effects within the Sizewell Marshes SSSI. This will have the consequence of potentially reducing or increasing flows to the Leiston Main Drain and therefore potentially interfering with water level control on Minsmere South Levels and potentially at the Minsmere Sluice. This does not appear to have been addressed and we can see no mitigation that achieves this function. In the absence of any further mitigation measures lower in the catchment, we cannot see how potential changes to water levels in the Minsmere to Walberswick SPA, Ramsar and SSSI site could be managed.

3.203 Section 1.3.19 (page 7) of the *Vol. 3: ES Addendum Appendices Ch. 2 Main Development Site Appendix 2.14.*[111] states:

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[107] ES Volume 2 Main Development Site Chapter 19 Groundwater and Surface Water Appendix 19B: Sizewell C Conceptual Site Model of the Hydrogeological Regime Part 1 (APP-304)

[108] ES Volume 2 Main Development Site Chapter 19 Groundwater and Surface Water Appendix 19F Sizewell Drain diversion outline design information (APP-309)

[109] A tilting weir is **a moveable weir used for raising and lowering a head of water by controlling the flow to a lower catchment area**. Typically, the plate or paddle of the tilting weir moves up and down in a narrow duct by titling, pivoting or rotating on its bottom horizontal axis - which opens or closes the tilting weir thus controlling the flow of water.

[110] A piped structure to control water levels consists of a tubular pipe with a 90 degree up-stand or non-return flap that allows water to be held back or discharged dependent upon how the pipe is operated

[111] Volume 3 Chapter 2 Environmental Statement Addendum Groundwater and Surface Water Appendices 2.14.A (AS-236)

'*It is envisaged that the principal mitigation options would relate to the new control structure to be installed at the northern end of the realigned Sizewell drain and operational practice within the Sizewell Marshes SSSI. Consequently, this approach is consistent with the existing operational management regime within the system.'*

3.204 Our concerns regarding the Leiston Main Drain above remain unanswered and the absence of any form of mitigation for effects transferred downstream also remains.

3.205 As mentioned above we have been unable to locate suitably detailed mitigation plans or a suitably robust approach to monitoring. We have covered this in further detail below. We do not concur that an approach '*allowing control measures to be implemented*' in response to '*unacceptable changes in flows and levels*' is sufficient to conclude that there will be no adverse effect on the interest features of the Minsmere to Walberswick SPA, Ramsar and SSSI site, especially nesting avocet and bittern.

Approach to Monitoring and Mitigation

3.206 Item MDS-GSW16 (page 26) in the *Mitigation Route Map Addendum* [112] identifies the

'repurposing the infiltration basin in WMZ5 to have a dual purpose of infiltration basin and water resources storage area enables the use of the previously identified water resources storage area for flood mitigation and wet woodland habitat creation. The change in use of the area from a water resources storage area to flood mitigation represents a change in a primary mitigation for the project'.

3.207 Whilst we welcome the provision of additional mitigation, we are not clear how the purpose of this additional storage at this location is intended to provide flood mitigation and what the intended benefit will be. We request more information to understand what elements of potential impact this mitigation is intended to address and how it will function to achieve this.

3.208 Section 1.1.2 (page 1) of *Vol. 3: Ch. 2 ES Addendum Appendix 2.14.A* [113] confirms

'This strategy [the Water Level Monitoring and Mitigation Plan] *sets out the principles of the monitoring, and the mitigation approach should monitoring identify that construction works within the main development site are leading to materially worse environmental effects on groundwater levels or quality at the site or surrounding area.'*

3.209 We welcome this intention.

3.210 Section 1.3.1 (page 3) of *Vol. 3: Ch. 2 ES Addendum Appendix 2.14.A* also states

'The assessment of potential changes to the water environment shows that the predicted changes are limited in extent, magnitude and duration such that no significant environmental impacts should occur. However, it is recognised that reassurance monitoring is required to demonstrate that the predicted change is realised, and not exceeded, as the project progresses.'

3.211 Again, we welcome this intention. However, given that we are not reassured that the primary mitigation measures are sufficiently progressed to satisfactorily support the predicted changes we regard it as essential to ensure that monitoring of the predicted changes and a further

[112] Mitigation Route Map (Consolidated) (REP1-006)
[113] Volume 3 Chapter 2 Environmental Statement Addendum Groundwater and Surface Water Appendices 2.14.A (AS-236)

adaptive mitigation strategy are required to ensure adequate protection of the Sizewell Marshes SSSI and Minsmere to Walberswick SPA, Ramsar and SSSI.

3.212 Section 1.2.5 (page 2) of *Vol. 3: Ch. 2 ES Addendum Appendix 2.14.A* describes the velocity and stage monitoring at the Applicant's monitoring location G8 – installed approximately 500m south of Minsmere Sluice in 2015 in order to monitor changes in flow and level in the Leiston drain downstream of the Applicant's monitoring location G1. This appears to be the only surface water monitoring point in or relevant to the Minsmere to Walberswick SPA, Ramsar and SSSI. Whilst we recognise that this is a key location to monitor any potential changes arising from the proposed development and their potential effects on the Leiston Main Drain, should it identify any changes then it will not be possible to assess any further potential impacts in other areas of RSPB Minsmere which could be affected by changes in the Leiston Main Drain, as described above.

3.213 We welcome the intention expressed in sections 1.3.2, 1.3.6 and 1.3.11 (pages 2 – 6) of *Vol. 3: Ch. 2 ES Addendum Appendix 2.14.A* to include sentinel boreholes to identify any potential changes that may extend to the Minsmere Walberswick Heaths and Marshes SSSI, committing to the monitoring focussing on the impact of the proposed development on the Sizewell Marshes SSSI and Minsmere Walberswick Heaths and Marshes SSSI. However, we would welcome the opportunity to understand the locations of these boreholes to ensure that these risks can be adequately monitored.

3.214 Section 19.5.8 (page 44) ES *Vol. 2 Ch. 19 Groundwater and Surface Water*[114] confirms some of the details discussed above including

- 'Sizewell Drain would be diverted
- At its northern extent, it would discharge to the Leiston Drain upstream of the SSSI crossing.
- revised water level management may be required for the drainage units and watercourses adjacent to the construction site, including water level control structures along the realigned Sizewell Drain and the revised operation of other existing structures.

3.215 To allow for *fine tuning of the management regime over time.* And to *prevent any detrimental impacts on groundwater from the Sizewell Drain.*

3.216 This paragraph also confirms that

> *The specific position, nature and operational parameters of the control structures will be determined in conjunction with stakeholders. This information could form part of an update to the Water Level Management Plan for the Sizewell Marshes SSSI*

3.217 With Section 19.5.12 (page 47) stating

> *'The permanent realignment of the Sizewell Drain could lead to a change in groundwater flow patterns, that could have an associated effect on the groundwater dependent ecological receptors (e.g. the Sizewell Marshes SSSI). To mitigate this, the realignment has been engineered to maintain a surface water level that will prevent alteration of the groundwater flow regime. Control structures would be installed along the realigned Sizewell Drain to manage water flow and prevent changes to groundwater. The specific position, nature and operational parameters of the control structures will be determined in conjunction with stakeholders.*

[114] ES Volume 2 Main Development Site Chapter 19 Groundwater and Surface Water (APP-297)

3.218 And will also form part of *an update to the Water Level Management Plan for the Sizewell Marshes SSSI,* confirming that the detail of the primary mitigation measures is yet to be determined and does not make it clear that this will be in place in time for the conclusion of the Examination.

3.219 We note in section 13.10 (page 5) of *Vol. 3: Ch. 2 ES Addendum Appendix 2.14.A* that

> *'the monitoring plan will be kept as a live document and revised regularly to ensure it remains fit for purpose for the activities being undertaken at the site. The monitoring plan, including any revisions, would be developed following consultation with East Suffolk Council, Suffolk County Council, the Environment Agency, Natural England and Suffolk Wildlife Trust, as appropriate.'*

3.220 We welcome the commitment to maintain a live document but question why the RSPB, as the main landowner in the adjacent sections of the Minsmere to Walberswick Heaths and Marshes SSSI and holding significant experience and expertise is not identified as a consultee. We also note the proposed role of the East Suffolk IDB in the document but note that they are not identified as a consultee for the monitoring plan.

3.221 Overall Vol. 3: Ch. 2 ES Addendum Appendix 2.14.A has no recognition that a mitigation plan is in place should the predicted change be exceeded in any circumstance beyond changes to water levels in the Sizewell Marshes SSSI.

3.222 No mitigation strategy is proposed to address changes to water levels in the Minsmere – Walberswick SPA, Ramsar and SSSI designated site and wider monitoring in this catchment beyond G8 and the sentient boreholes (at unspecified locations) is not presented. There is a lack of clear framework throughout the document. It proposes the monitoring plan will inform a Water Level Management Plan prepared and owned by the IDB (1.3.2, p3), but then describes a monitoring strategy in section 1.3.6 (p5) where a framework of 'trigger' and 'action' levels may be captured in an updated Water Level Management Plan. Section 1.3.14 and 1.3.15 (p6) proposes an adaptive trigger level approach to mitigation which would be implemented as a standalone control measure but could be used by the IDB to inform any update to the Water Level Management Plan for Sizewell Marshes SSSI not allowing for a clear understanding of the intended approach.

Potential impacts of water levels on the Minsmere to Walberswich SPA, Ramsar site and SSSI

3.223 Table 5.1 (page 18) of ES*Vol. 2 Ch. 19 Appendix 19B: Sizewell C Conceptual Site Model of the Hydrogeological Regime*[115] assess the ecological receptors and their hydrological sensitivity for the adjacent SPA using wetland habitats located within South Minsmere Levels (south of the new cut) and Sizewell Marshes. It proposes a medium hydrological sensitivity for breeding waders (but depends on how important groundwater is in the maintenance of damp ditch edges during the breeding season i.e. April to June) and a low sensitivity for breeding and wintering wildfowl and foraging marsh harrier and bittern.

3.224 Whilst we agree with the Applicant's approach for the sensitivities to species (and the species' sensitivities) using the South Levels, the assessment does not take into account the hydrological sensitivity of the SPA species utilising habitats north of the New Cut which could be affected by increased flows to the Minsmere Sluice.

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[115] ES Volume 2 Main Development Site Chapter 19 Groundwater and Surface Water Appendix 19B: Sizewell C Conceptual Site Model of the Hydrogeological Regime Part 1 (APP-304)

3.225 We therefore believe the assessment of impacts on SPA bird species in hydrological models has not taken account of all potential impacts.

Summary of Hydrological Concerns affecting Minsmere

Surface Water Levels conclusions

3.226 The assessment of no likely significant effect relies on a series of proposed primary mitigation features which have yet to be defined in sufficient detail as to confirm that they will be effective. Without this assurance of effective embedded mitigation we cannot support the assessment conclusion. A monitoring strategy to assess whether predicted changes are exceeded is proposed. However, we do not believe the proposed monitoring is sufficient and the mitigation strategy to address any change to predicted levels is lacking. In addition, the geographical scope of the monitoring and mitigation strategy does not adequately address the risks to designated interest features in the Minsmere to Walberswick SPA.

3.227 We note that our concerns are reflected by the Secretary of State's 2019 Scoping Opinion on the project. Scoping opinion[116] paragraph 4.11.8 states *Paragraph 6.12.28 of the Scoping Report states that "a control structure" is proposed in the realigned Sizewell Drain. No further information regarding the "control structure" is included in this chapter or in the project description in the Scoping Report. The ES should describe the control structure in the project description including where it will be located, how it will operate, and include environmental assessments where relevant within the technical assessments.* The Applicant has responded[117] *The assessment of the likely effects associated with the realignment of drains, including the incorporation of a new control structure, is addressed in the relevant ES chapter (Volume 2, Chapter 19).* However, as we have highlighted, the Applicant has yet to describe where the control structure will be located and how it will operate and without the details requested, we cannot ascertain its likely effectiveness.

3.228 We are disappointed to have reached this stage of the Application process with so much required information still missing.

Hydrological Impacts and Risks to Sizewell Marshes SSSI.

Sizewell Marshes

3.229 As set out in more detail above Sizewell Marshes SSSI is situated to the west of Sizewell nuclear power station and is a complex of marshes with a unique mix of geology and hydrology that has created nationally significant botanical and invertebrate diversity. The value of the SSSI is enhanced by its close proximity and connectivity to RSPB Minsmere. The site has three main habitats;

- species rich grazing marsh - floristically, the grazing marshes and its network of dykes is one of the best examples of this habitat remaining in East Anglia. The dominant vegetation group is M22 fen meadow, with intermediate areas between this dominated by MG10 rush pasture. The M22 communities are very species rich and contain a wide range of plants of conservation interest. The dyke system supports a wealth of plant species including nationally scarce species
- tall herb fen; and

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[116] From page 56 of the Scoping Opinion for Proposed Sizewell C Nuclear Development. The Planning Inspectorate, July 2019

[117] From page 35 of ES Volume 1 Introduction to the Environmental Statement Chapter 6 EIA Methodology Appendix 6C - Responses to EIA Scoping Opinion Comments

- wet alder carr woodland - also supporting a very high invertebrate diversity.

3.230 In addition

- 18 species of dragonfly have been recorded including rarities such as Norfolk hawker (*Aeshna isosceles*), hairy dragonfly (*Brachytron pratense*) and lesser red eyed damselfly (*Erythromma viridulum*).
- 236 moth species have been recorded since 1996 including nationally rare species.
- 12 species of grasshopper and cricket; a good range of spiders, including the nationally rare *Leiobunum rotundum*; and three nationally rare species of fly
- 148 bird species have been recorded with a total of 80 breeding species including little egret, gadwall, kingfisher, barn owl & water rail and Cetti's warbler and wintering birds include bittern, bearded tit and snipe.
- Mammals include otters, water voles, water shrews and harvest mouse as well as red deer.
- Bats - the SSSI is thought to hold an excellent bat community including barbastelle, Natterer's, all three pipistrelle species, brown long eared, noctule, serotine, and Daubenton's recorded annually. It is likely that nationally rare barbastelle and regionally scarce Natterer's bat breed on site.

3.231 We are grateful to Friends of the Earth and their Experts (Dr Rob Low, Dr David Mould and Jon Graham) for considering these issues in detail in their written representations: "A critical review of Sizewell C Co's site characterisation, impact assessment, and proposals for impact mitigation, in relation to risks posed to the ecohydrological integrity of Sizewell Marshes SSSI."

3.232 We refer the Examining Authority to those representations for full consideration of the issues summarised below.

Data Adequacy

3.233 The omission of a comprehensive surface water and shallow groundwater monitoring programme has prevented the development of a satisfactory conceptual understanding of the fine-scale ecohydrological dynamics of the system. Such an understanding is critical for an effective assessment of risk;

3.234 The hydro(geo)logical functioning of the shallow zone within Sizewell Marshes SSSI, which controls the variables which define the hydrological supporting conditions for the M22 fen-meadow, should have been monitored, analysed and characterised in much more detail, using a contemporary ecohydrological approach;

3.235 For example, fully-screened shallow (c. 1 m deep) dipwells should have been installed in the Peat at the most notable stands of M22 (e.g., FM2 and FM3c, see Section 4.3), in order to allow assessment of the relative contribution of groundwater discharge as a water source at these points;

3.236 The water table elevation in the Peat is never plotted in relation to the ground surface. This variable is very widely recognised as the most important in relation to defining hydrological supporting conditions for M22;

3.237 The absence of rainfall data, which obviously represents a primary control on surface water and groundwater behaviour, will have compromised the analysis of all other hydrological responses at a fundamental level;

3.238 By adopting a more comprehensive approach, it may have been possible to identify the actual, rather than the 'likely' mechanisms which explain the expression of M22, and therefore to develop a more detailed and certain ecohydrological conceptual model;

3.239 Where the qualitative assessment concluded that a potential impact was not significant, further detailed analyses were not undertaken. This is a process failure, not following the precautionary principle, to account for potential errors in the qualitative assessment.

Interpretation of the data and the numerical model

3.240 The ecohydrological analysis and conceptual model presented by EDF has failed to identify the controlling variables and mechanisms which directly control the variables defining the hydrological supporting conditions for the M22 community within Sizewell Marshes SSSI;

3.241 Paragraph 19.4.26 in APP-297 states that '*Groundwater levels within the Peat Deposits were noted to be highly responsive to temporary pumping during maintenance works at the Minsmere Sluice from October 2013 to February 2014 suggesting a high degree of connectivity between the surface water network and the Peat groundwater system*'. This high degree of connectivity fundamentally undermines the qualitative assessment, as it shows that changes to water levels in one part of the site will induce a change in other parts. As demonstrated below, the impacts of these changes are significant;

3.242 Projected drawdown could easily take the water table elevation outside of the optimal range for the summer water table, and therefore shows that the M22 community and key aquatic plants within the associated ditch system are significantly vulnerable to the projected drawdowns. This is not considered as anything more than a 'not significant' risk to the designated site, which is a major failing of the risk assessment process;

3.243 Consideration of only a doubling in hydraulic conductivity is unrealistically optimistic, and that the possibility that the overall hydraulic conductivity would be three- to five-times higher than designed should be assessed. If the directly proportional relationship between hydraulic conductivity of the cut-off wall and water table drawdown is assumed, this would give predicted drawdowns of the summer water table in the region of 30-50 cm. M22 is clearly extremely sensitive to this magnitude of drawdown.

Ecological relevance of Groundwater to M22 Fen Habitat

3.244 Upward groundwater flow and discharge, in response to the hydraulic gradient from the Crag to the Peat, is almost certainly a critical source of water to some of the stands of M22;

3.245 Most stands of M22 are associated with permanent or intermittent seepages or where the water table is shallowly subsurface all year, sometimes peripheral to permanent seepages. This indicates a strong preference for soligenous wetness, where wet conditions are maintained by continuous incoming flow, rather than downstream (usually topographic) impoundment;

3.246 M22 is typically found is base-rich conditions over a wide range, but usually with a moderate level of fertility. Some of the least fertile sites were the most species-rich.

Impact Pathways on the M22 Fen Habitat

3.247 As a consequence of the critical role of groundwater, there are three major developments that have the potential to have a major impact on the sensitive ecohydrological functioning of the wetland system. These are:

1. The new cut-off wall adjacent to the platform base for the new site, and related internal lowering of groundwater levels (dewatering) during the construction phase;

2. The re-routing of the Sizewell Drain; and

3. The new crossing at the drainage outlet for the Sizewell Drain and the Leiston Drain.

3.248 Changes to the wetland system will have unknown impacts given the lack of understanding of how the system is operating at the fine scale, as noted in the section above. This is compounded by a lack of detail on the design of the three main critical developments;

3.249 Sizewell Marshes SSSI nitrate thresholds are close to UKTAG (2014) levels for M22 fen. Irrigation of the M22 fen-meadow vegetation with surface water i.e. the proposed mitigation, is therefore likely to cause further damage;

3.250 Paragraph 1.3.33 (Appendix 19E, APP-309) notes that '*the consented discharge from Leiston STW contains higher levels of nutrients (including phosphates) than the rest of the drainage system*', and that '*in order to reduce the ingress of nutrients into the wider drainage network, SWT (Suffolk Wildlife Trust) have attempted to reduce connectivity between Leiston Drain and the drainage network on either side (Sizewell Belts and Sizewell Marshes) via the use of flow regulating structures'*. It is of significant concern to note that the proposed mitigation strategy appears to reverse this established protective practice;

3.251 M22 is strongly associated with soligenous conditions (see Section 4.2), and therefore fostering of more topogenous conditions is completely inappropriate;

3.252 A wider span bridge design would be far less impacting, as it would leave a significantly wider corridor unchanged, to facilitate original hydraulic control and subsurface drainage of the SSSI site.

Monitoring and mitigation plan

3.253 It is important to note that a monitoring plan does not yet exist. The description of the proposed monitoring plan within the strategy document is insufficiently detailed;

3.254 The proposed in-field monitoring programme across the Sizewell Belts appears to have been poorly designed and not able to facilitate an appropriately detailed understanding of the hydrological dynamics of the wetland system.

3.255 The implied assumption that the development of a hydro(geo)logical 'monitor and mitigate' plan with stakeholders will be relatively straightforward and to schedule is completely unrealistic. Development and agreement of the monitoring and mitigation plan, since it forms a part of the ecohydrological viability of the proposed development, should therefore be addressed at the earliest stage in the planning process.

Summary and conclusions regarding Sizewell Marshes

3.256 From the available evidence, Friends of the Earth's Experts have concluded that direct, upward groundwater flow and discharge, in response to the hydraulic gradient from the Crag to the Peat, is almost certainly a critical source of water to some of the stands of M22. This is critical because it allows favourable hydrological supporting conditions to be maintained, in terms of water table elevation regime and water quality, to enable the favourable condition of the SSSI to be maintained.

3.257 Our view is that EDF's conceptual model is significantly flawed and therefore that the prediction of impacts (including the design and use of the numerical model) has not been informed to the best possible degree.

3.258 Development and agreement of a detailed monitoring and mitigation plan, since it forms a part of the ecohydrological viability of the proposed development, should be addressed at the earliest stage in the planning process. The required contents of the plan, such as the scope of monitoring, data quality-checking procedures and reporting, detailed actions and timescales in relation to the loss of a monitoring point, periodic reporting requirements, and the requirements of the stakeholders which review the reports, all have long-term cost and logistical implications for the developer and should be agreed before determination.

3.259 The primary measure for mitigation of ecohydrological impacts from the development appears to be that, if drawdown of the water table in the Peat within Sizewell Marshes is larger than predicted, water levels in the SSSI drainage ditch network will be raised, such that water migrates from the ditches into the Peat layers to maintain in-field water table elevations. This measure is fundamentally inappropriate and would actually cause further damage to the M22 within the SSSI. This is because the historical recorded nutrient concentrations within the ditch network indicate that the threshold value for potential damage of mesotrophic and fen-meadow fens would frequently be exceeded. This is unsurprising as a significant percentage of incoming flow comes from the Leiston STW.

Final summary regarding Sizewell Marshes

3.260 The Applicant's understanding of the environmental processes which support the nationally important M22 fen community and associated communities within Sizewell Marshes SSSI is flawed, because up-to-date ecohydrological knowledge and techniques have not been applied. This has led to ill-informed impact prediction, which has resulted in the likelihood, magnitude and significance of potential impacts being significantly underestimated. These problems have been compounded by the Applicant's proposal of a mitigation technique which would actually cause further damage to the SSSI, rather than mitigating any unexpectedly large impacts.

Noise and visual disturbance (birds)

Background

3.261 Our main concerns in relation to noise and visual disturbance relate to the construction phase of the development and in particular to the main development site and impacts on bird features of the Minsmere-Walberswick SPA and Ramsar site (designated sites) and the Sizewell Marshes SSSI[118]. Construction activity may result in disturbance and/or displacement of birds, through responses to visual or acoustic stimuli, and the level and type of sensitivity varies by bird species and by season. In brief, disturbance as a result of construction activity may affect birds by causing:

- reduced food intake rates
- increased energy expenditure
- decreased breeding productivity
- physiological changes
- habitat loss due to displacement

3.262 Please note that birds generally use the most suitable habitats available to them at any point in time, therefore displaced birds are likely to be pushed into less suitable habitats, meaning

[118] See ES Vol. 2, Ch. 14 Terrestrial Ecology, App 14A2 - Designated Sites Annex 14A2.1: Figures 14A2.1 - 14A2.3 (APP 228)

displacement itself may also lead to other impacts listed above, with effects on reproductive success potentially leading to a population decrease (Alves *et al*., 2014[119]).

3.263 The evidence base for the Applicant's assessment of noise impacts is set out in section 8.8 b) iv. of the Shadow HRA Report Vol. 1 Part 1[120] while ES Vol. 2 Ch. 11 Noise and Vibration[121] section 11.3 d) describes the scenarios assessed as part of the noise modelling. It explains the types of activities planned to take place over the five construction phases during both the daytime and night-time construction shifts. It is stated that construction Phase 1 (site establishment and preparation for earthworks) and Phase 5 (removal of temporary facilities/restoration of the land) will result in the highest noise levels, although it is also noted that the construction phases will overlap. The outputs of the noise modelling relevant to important ecological receptors are presented as noise contour maps and are found in the Shadow HRA Report Vol. 1 Part 3[122] with updates in the Shadow HRA Report Addendum Appendices 1A-10A Part 5[123].

3.264 The evidence base for the Applicant's assessment of visual disturbance impacts is also set out in section 8.8 b) iv. of the Shadow HRA Report Vol. 1 Part 1. Again, concerns primarily relate to the construction period and in particular to lighting of the construction site, however, the presence of human activity and large or moving machinery/infrastructure on the skyline may also have an effect. In general, the predicted impacts of visual disturbance are encompassed within the areas predicted to be affected by noise disturbance, although a few exceptions are noted and discussed further below.

Sites and bird species of concern

3.265 Our concerns primarily relate to potential impacts on breeding marsh harrier and breeding and non-breeding waterbirds of the Minsmere-Walberswick SPA and Ramsar site and on two sites functionally linked to the SPA: Minsmere South Levels (part of Minsmere-Walberswick Heaths and Marshes SSSI) and Sizewell Marshes SSSI.

3.266 The Minsmere-Walberswick SPA[124] was classified in 1991 and contains extensive areas of freshwater and coastal grazing marsh, coastal reedbeds, saltmarsh, lowland heathland, woodland, intertidal mud and mixed sediment. It supports the following qualifying features:

- Avocet (*Recurvirostra avosetta*), Breeding
- Bittern (*Botaurus stellaris*), Breeding
- Gadwall (*Mareca strepera*), Breeding
- Gadwall (*Mareca strepera*), Non-breeding
- Greater white-fronted goose (*Anser albifrons albifrons*), Non-breeding
- Hen harrier (*Circus cyaneus*), Non-breeding
- Little tern (*Sternula albifrons*), Breeding
- Marsh harrier (*Circus aeruginosus*), Breeding
- Nightjar (*Caprimulgus europaeus*), Breeding
- Shoveler (*Spatula clypeata*), Breeding
- Shoveler (*Spatula clypeata*), Non-breeding
- Teal (*Anas crecca*), Breeding

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119 Alves, M., Ferreira, J., Torres, I., Fonseca, C. & Matos, M. (2014) Habitat Use and Selection of the Marsh Harrier *Circus aeruginosus* in an Agricultural-Wetland Mosaic. Ardeola. 61(2): 351-366

120 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 (APP-145)

121 ES Vol. 2 Main Development Site Chapter 11 Noise and Vibration (APP-202)

122 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 3 (APP-147)

123 Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)

124 The SPA designation - http://publications.naturalengland.org.uk/file/5944954236960768

3.267 Natural England has set site conservation objectives for the Minsmere-Walberswick SPA[125] as follows:

> "*The objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:*
>
> - *the extent and distribution of the habitats of the qualifying features*
> - *the structure and function of the habitats of the qualifying features*
> - *the supporting processes on which the habitats of the qualifying features rely*
> - *the populations of each of the qualifying features*
> - *the distribution of qualifying features within the site*"

3.268 The Minsmere-Walberswick Ramsar site[126] was designated in 1976 under:

- Ramsar criterion 1 for containing a mosaic of marine, freshwater and marshland with a rare transition in grazing marsh ditch plants from brackish to fresh water and;
- under criterion 2 for supporting nationally scarce plants, British Red Data Book invertebrates and nationally important numbers of breeding and wintering birds (bittern, gadwall, shoveler, teal, marsh harrier, avocet and bearded tit).

3.269 The Minsmere South Levels form part of the Minsmere-Walberswick Heaths and Marshes SSSI[127] which comprises a suite of habitats including mudflats, shingle beach, reedbeds, heathland and grazing marsh. It is designated for its breeding and wintering birds, scarce plants and important invertebrate populations.

3.270 Sizewell Marshes SSSI128 contains lowland, unimproved wet meadows which support outstanding assemblages of invertebrates and breeding birds along with ditch systems, reedbeds and fens. Several nationally scarce plants are also present.

Comments on the baseline data

Breeding bird survey methodology

3.271 Paragraph 3.2.8 of the Breeding Bird & Waterfowl Survey Report 2020[129] explains that breeding waders and waterfowl were surveyed on the Minsmere South Levels in 2020 from vantage points. This is a departure from standard methodologies which require a walked transect route, such as those carried out by the RSPB. Vantage point surveys are likely to underestimate breeding activity for waders and wildfowl, particularly given the viewing distances involved. Whilst there were understandable constraints on surveying in 2020 due to covid restrictions, this potential for underestimation (and potential lack of comparability between surveys in different years) should be acknowledged and taken into account. We note that distributional data for breeding birds were not collected by the Applicant prior to 2020, and whilst figures for breeding birds have been obtained from the RSPB, the lack of longer-term distributional data (following a standard repeated visit methodology and usually for a minimum of two years) represents a significant limitation to the Applicant's impact assessment.

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125 The SPA generic Conservation Objectives - http://publications.naturalengland.org.uk/file/4811128974868480

126 The Ramsar designation - https://jncc.gov.uk/jncc-assets/RIS/UK11044.pdf

127 Minsmere-Walberswick Heaths and Marshes SSSI - https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1000721.pdf

128 Sizewell Marshes SSSI - https://designatedsites.naturalengland.org.uk/PDFsForWeb/Citation/1003416.pdf

129 Breeding Bird & Waterfowl Survey Report 2020 in the Additional Ecology Baseline Survey Reports November 2020 (AS-021)

Comments on the assessment methods for potential noise and visual effects on birds during construction

Comparison of predicted noise levels with those recorded at Minsmere

3.272 Shadow HRA Report Vol. 1 Part 1[130] paragraphs 8.8.185-186 refers to a survey of noise levels at RSPB Minsmere conducted near South Hide which is used to assess noise levels experienced by breeding birds. The noise effects described include anthropogenic sources including motor vehicles and visitors, resulting in maximum sound levels of over 70dB.

3.273 We note the acknowledgement that the acoustic data collected at Minsmere were from an area close to visitor routes and may not be representative of the wider area. South Hide is on a visitor route between the main scrape and the eastern end of the reedbed and is therefore not appropriate for assessing the noise levels likely to be experienced by most breeding birds – it would not be expected that visitors and motor vehicles would result in the noise levels quoted 'at the bird' given the distance of the birds from these sources. It should also be clarified as to whether any atypical management work was being carried out at that location during the survey period.

3.274 With regard sound levels attributed to insect noise and birdsong, it is important to draw a distinction between anthropogenic noise which is likely to cause alarm to birds and noise sources that are natural (e.g. birdsong, the sea) and hence much less likely to cause disturbance.

Definitions of daytime and night-time

3.275 Paragraph 1.2.2.2 of ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment[131] explains that assessment scenarios have been defined separately for daytime (07:00–23:00hrs) and night-time (23:00–07:00hrs) construction. However, this means that "daytime" will include a significant proportion of dark hours particularly in winter. With full construction activities ongoing until 11pm, we are concerned regarding the potential for ecological impacts of noisy activities during dark hours – e.g. potential displacement of birds arriving to roost. Impacts under the daytime scenario could therefore be greater than predicted.

3.276 For ecological receptors 'day' is dawn to dusk and 'night' is dusk to dawn. Assessing impacts during those time periods would therefore be more relevant.

Overlapping construction phases

3.277 Plate 3.1 in ES Vol. 2 Ch. 3 Description of Construction[132] indicates that that the main construction phases are expected to overlap. The outputs of the noise modelling are presented for each construction phase and it is not always clear whether these overlaps have been accounted for. Combined worst case scenario (WCS) maps where construction phases are expected to overlap and impacts would be worse than each construction phase treated alone should be provided where necessary to avoid under-estimation of potential impacts.

Assumptions regarding construction timelines

3.278 Paragraphs 8.8.190 – 202 of the Shadow HRA Vol. 1 Part 1[133] provide a more detailed breakdown of noise effects by construction phase, and assert that, as most noisy activities will not occur at same time, noise impacts are overestimated in peak noise levels and duration. Construction

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130 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
131 ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204)
132 ES Vol. 2 Main Development Site Ch. 3 Description of Construction (APP-184)
133 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

timelines at weekly/monthly scales are highly likely to be subject to change, and as acknowledged in paragraph 1.3.1.3 of ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment[134], the final construction timelines will only be confirmed once contractors have been appointed meaning that some changes would be expected.

3.279 We therefore do not agree that sufficient reliance can be placed on these breakdowns to assume the results are over-precautionary. The WCS is the only safe scenario for assessment.

Assumption that acoustic fence will be in place during Phase 1 of the construction period

3.280 Paragraph 8.8.67 of the Shadow HRA Report Vol. 1 Part 1[135] explains that the impact assessment for the Phase 1 construction period is based on the assumption that a 5m high fence (or equivalent mitigation) will be in place to reduce potential noise impacts. It is crucial that the 5m acoustic fence is constructed at the beginning of the Phase 1 construction period, especially as Phase 1 of the construction period is one of the phases with the highest noise impacts on sensitive areas[136]. If there is a period where construction is taking place without mitigation in place, the assessment of Phase 1 impacts could be under-precautionary.

Noise modelling limitations

3.281 Section 1.3.3 of ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment[137] explains the assumptions and limitations behind the noise modelling. There are several activities where sufficient details are not currently available to properly model resultant noise. These include early morning unloading of HGVs, night-time dewatering activities and marine unloading. It is noted that these issues will require additional modelling once further details are known and therefore, we consider that it is possible that some impacts may have been underestimated at this stage.

Uncertainty regarding noise impacts during construction Phase 5

3.282 Section 1.3.3 of ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment[138] explains that limited details were available regarding the removal of temporary facilities and land restoration during construction Phase 5 and therefore that indicative average noise levels are presented for this Phase. Paragraph 1.3.4.3 explains that short term noise levels during Phase 5 are likely to be similar to those of Phase 1 but that:

> *"it is not currently possible to define the duration or activities during this period (or associated noise predictions) based on the limited assumptions presented for Phase 5".*

3.283 Also, paragraph 1.3.4.4 acknowledges that:

> *"short periods of higher sound levels can be expected when restoration works (which may include breaking up of hardstanding areas) are taking place at the edges of the site nearest receptors".*

3.284 The potential for high noise levels during this Phase needs to be recognised throughout the assessments as in many instances Phase 1 is referred to as the worst case for ecological impacts of noise. Given that Phase 5 spans the final two to five years of construction (noting inconsistency regarding phase length between documents) this would significantly increase the duration of

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134 ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204)
135 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
136 See page 17, Figure 8A.8 in Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)
137 ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204)
138 ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204)

major noise impacts. Also, the uncertainty in these predictions should be clarified and any resulting differences in how the figures presenting the noise contours for each Phase should be interpreted should be explained.

3.285 In addition, the figure for Phase 5 (Figure 8.6 in Shadow HRA Report Vol. 1 Part 3[139]) appears to be missing the noise contour representing 64-70dB, making it impossible to identify the area subject to potential impacts in this Phase (no figure for Phase 5 is presented in the updated noise modelling in Shadow HRA Report Addendum Appendices 1A-10A Part 5[140]).

Chronic noise modelled for Construction Phases 3 and 4 only

3.286 Paragraph 1.2.9 of the Shadow HRA Report Addendum[141] explains that additional modelling of chronic noise levels is presented for Phases 3 and 4 of the construction period. Whilst we welcome the provision of this information, based on the original modelling of impulsive noise levels, it appears that Phases 3 and 4 are overall the least noisy construction phases. Although the Shadow HRA Report Addendum states that these phases were chosen as they are the longest construction phases and hence would represent the longest period of impact, we query why Phases 1, 2 and 5 are not included as noise levels might be expected to be higher during these phases.

Noise and visual disturbance evidence base

Impulsive noise disturbance threshold

3.287 We have in previous consultations noted that we have some reservations around the use of a 70dB L_{Amax} threshold for impulsive noise impacts on marsh harrier (please see below regarding specific impacts on marsh harriers), although we also acknowledged that there was limited evidence on which to challenge this. These reservations also apply to waterbirds. We acknowledge that a lower threshold of 65dB L_{Amax} has been applied for this assessment of impacts on breeding waterbirds, while the 70dB L_{Amax} threshold is used for wintering waterbirds.

3.288 We also acknowledge the discussion of the evidence for noise impacts on waterbirds in the Shadow HRA Report although we note that much of this evidence relates to impacts on estuarine birds in winter and that limited evidence exists regarding impacts from construction noise specifically. We therefore reiterate the importance of robust and demonstrably successful mitigation for any impacts.

Temporary effects of noise disturbance

3.289 Paragraph 8.8.33 of the Shadow HRA Report Vol. 1 Part 1[142] states that:

> *"Evidence from studies into this type of noise pollution [infrequent intense noise] indicates that generally birds, if undertaking a flight response, will return to the affected area once the noise has dissipated."*

3.290 One of the references provided for this statement applies to a study of wintering waterbirds on a field near the Humber Estuary (Wright *et al*., 2010)[143] and the other refers to a study of the

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139 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 3 (APP-147)
140 Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)
141 Shadow HRA Report Addendum (AS-173)
142 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
143 Wright, M.D., Goodman P. & Cameron. T.C. (2010) Exploring behavioural responses of shorebirds to impulsive noise. Wildfowl 60: 150-167

Mexican spotted owl (Delaney *et al.*, 1999)[144]. The statement is part of the noise disturbance evidence base for breeding and non-breeding bird species. We consider that this statement is too general to be of use in an assessment such as this as there are a wide variety of responses exhibited by birds of different species, in different habitats and with varying life histories and at different seasonal stages; any such generalisations should where possible be taken from species with characteristics in common with the species of concern here. If such evidence is not available, the limitations of any conclusions drawn should be acknowledged. We also consider that where possible, evidence around disturbance and displacement effects should arise from similar causes, although we acknowledge that evidence relating to construction specifically is limited.

3.291 It should also be noted that the statement does not indicate how long it may take for birds to return following the removal of the source of disturbance, or whether desertion may eventually occur if infrequent intense noise occurs over a long period.

Generalisations regarding impacts of noise

3.292 As discussed above, the noise disturbance evidence base relates to a range of different species with different life histories and sensitivities. It is presented in the Shadow HRA Report section for breeding avocet, but then referred to in assessments for other species. We acknowledge the limited research evidence available for the individual species covered by the Shadow HRA Report but we recommend that where evidence is available for the species in question or a closely related species, that this is discussed in the individual species accounts rather than applying the generic evidence base to all species.

Lack of sensitivity threshold for chronic noise

3.293 Paragraphs 8.8.42-50 of the Shadow HRA Report Vol. 1 Part 1[145] discusses a selection of studies that have shown that chronic noise disturbance has the potential to affect densities of breeding birds. Effects can include:

- the masking of acoustic communication,
- effects on detection of predators
- the ability to hunt

3.294 Such impacts can result in changes to bird abundances and distributions and effects on adult and nestling fitness.

3.295 The Shadow HRA Report goes on to suggest sensitivity thresholds for impacts on both breeding and wintering birds, however thresholds are only suggested in relation to impulsive noise (as it is stated in paragraph 8.8.51) which is considered more likely to result in a behavioural response. Based on the evidence around breeding birds, we disagree with this statement and note that as chronic noise has been shown to affect densities and distribution of breeding birds, impacts of chronic noise could have a direct effect on the conservation objectives of designated sites.

3.296 We also note that impacts of chronic noise on birds have been shown even at low levels, although we acknowledge the limited number of studies available which include figures for chronic noise levels and the lack of evidence regarding the particular species of interest here. Senzaki *et al.* (2016)[146] studied the effects of traffic noise on foraging short-eared owls *Asio flammeus* and long-eared owls *Asio otus*. They reported that "Owls' ability to detect prey was affected even at the

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144 Delaney, D.K., Grubb, T., Beier, P., Pater, L. & Hildegard Reiser, M. (1999) Effects of helicopter noise on Mexican spotted owls. Journal of Wildlife Management, 63(1): 60-76.

145 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

146 Senzaki, M., Yamaura, Y., Francis, C.D. & Nakamura, F. (2016) Traffic noise reduces foraging efficiency in wild owls. Scientific Reports 6: 30602.

lowest level of [traffic noise] (40 dB[A]) and was approximately 17% lower than that of ambient conditions". McClure *et al.* (2013)[147] used speakers to create a 'phantom road' to experimentally test the effects of road traffic noise on bird distributions. They found over a one-quarter decline in bird abundance along the 'phantom road' and almost complete avoidance by some species, compared to control conditions. They also found negative relationships between bird abundance and average noise levels (which ranged from approximately 37 to 57 dB(A)) for over half the species studied. Therefore, based on the limited evidence available, we consider that some level of impact may be likely above noise levels of around 45dB L_{Aeq}.

Visual disturbance

3.297 As discussed for noise disturbance, much of the research evidence presented in the visual disturbance evidence base in the Shadow HRA Report relates to impacts on intertidal estuarine habitats, which may not be directly comparable to the species and habitats under consideration here. It is also notable that evidence relating to disturbance specifically from construction is also limited, as acknowledged in the evidence base.

3.298 Therefore, much of the evidence relating to non-lighting impacts is around recreational activities on estuaries. Given the lack of evidence around the specific impacts under consideration here, we are concerned that in many areas the proposed visual disturbance buffer of 300m has been further reduced to 150m due to the presence of natural screening. We are concerned that, due to the height of some of the infrastructure, cranes etc involved and the lighting of the site, the screening will not reduce all potential visual impacts.

Summary of extent of noise and visual disturbance impacts on birds during construction

Daytime impulsive noise

3.299 This discussion of daytime (07:00 – 23:00) impulsive noise levels is based on the modelling outputs presented in Figures 8.2 – 8.6 in the Shadow HRA Vol. 1 Part 3[148] and the updated modelling presented in Figures 8A.1 – 8A.3 and 8A.8 – 8A.9 of the Shadow HRA Addendum Appendices 1A-10A Part 5[149]. Paragraph 8.8.76 of the Shadow HRA Vol. 1 Part 1[150] explains that:

> "*Encroachment of the 64dB L_{Amax} footprint onto both areas [South Levels and Sizewell Marshes] is substantive during Phases 1 and 2… A large part of the Sizewell Marshes continues to be encompassed by the 65dB$_{LAmax}$ contour during Phases 3 and 4, but by Phase 5 there is little encroachment…*"

3.300 The encroachment of the 65dB L_{Amax} contour shown in the updated modelling outputs for Phase 2 (Fig. 8A.9) is greater still, noting that the update for Phase 1 only applies to the winter period (see Shadow HRA Report Addendum Appendices 1A-10A Part 5, and our concerns noted above should this scenario extend into the breeding season).

3.301 With regard to the stated encroachment for Phase 5, this is not clear from the original figure referred to as a contour is missing from the map (see comments above) and no updated modelling is provided for Phase 5 – we therefore request that evidence is presented to support this statement as based on the map, it appears that impacts could be significant.

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147 McClure, C.J.W., Ware, H.E., Carlisle, J., Kaltenecker, G., & Barber, J.R. (2013) An experimental investigation into the effects of traffic noise on distributions of birds: avoiding the phantom road. Proceedings of the Royal Society B 280: 20132290.

148 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 3 (APP-147)

149 Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)

150 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

3.302 Based on the noise impacts of Phases 1 to 4 only (given the uncertainty regarding Phase 5), this equates to potentially significant noise impacts on breeding birds of the Minsmere South Levels for the first 4-5 years of construction and to those of Sizewell Marshes for first 10-11 years. Should impacts during construction Phase 5 also be significant then breeding birds on Sizewell Marshes would be affected throughout the entire construction period of 12 years and on the Minsmere South Levels for up to 9 ½ years (see 'Background' section above for potential effects of disturbance and displacement on birds).

3.303 As noted above, updated modelling has been provided for the first winter of Phase 1 to incorporate the construction of the flood compensation and wetland area in the northern part of the construction area and the construction of the second Beach Landing Facility (BLF). This has the effect of extending the area of the Minsmere South Levels affected by the 70dB L_{Amax} noise contour (in relation to wintering bird species), to include parts of the southern and western areas of the Levels and a small area of the Minsmere reedbeds. Parts of the eastern block of Sizewell Marshes and the western tip are also within the 70dB L_{Amax} contour during this period, extending potential impacts to birds species into those additional areas.

3.304 The areas affected by the 70dB L_{Amax} contour during Phase 2 are smaller than those in Phase 1 but with significant parts of the eastern area of Sizewell Marshes still affected. During Phases 3 and 4 the encroachment reduces further still.

3.305 As discussed above, no updated modelling is provided for Phase 5 and the original outputs are missing a key contour, hence impacts during this phase are unclear but appear to be significant.

Daytime chronic noise

3.306 This discussion of daytime chronic noise levels is based on the updated modelling presented in Figures 8A.4 of the Shadow HRA Report Addendum Appendices 1A-10A Part 5[151]. Paragraph 8.3.14 of the Shadow HRA Report Addendum[152] explains that:

> *"In terms of chronic noise during the daytime period, predicted levels are lower than for impulsive noise during the same construction phase, as would be expected... For Phases 3 and 4, predicted noise levels are relatively low (at 40 - 50dB L_{Aeq}) across much of the Minsmere South Levels but higher across the Sizewell Marshes (where much of the eastern parts are encompassed by values above 55 dB L_{Aeq})."*

3.307 We are concerned that during Phases 3 and 4 much of Sizewell Marshes and Minsmere South Levels will experience chronic noise levels of above 40-45dB L_{Aeq} and could therefore experience impacts on breeding birds. As discussed above, we are also concerned that chronic noise levels for Phases 1, 2 and 5 are not included as noise levels might be expected to be higher during these phases.

Night-time chronic and impulsive noise

3.308 This discussion of night-time (23:00 – 07:00) chronic and impulsive noise levels is based on the updated modelling presented in Figures 8A.5 and 8A.6 of the Shadow HRA Report Addendum Appendices 1A-10A Part 5[153]. Paragraph 8.3.15 of the Shadow HRA Report Addendum[154] explains that:

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151 Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)
152 Shadow HRA Report Addendum (AS-173)
153 Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)
154 Shadow HRA Report Addendum (AS-173)

"Night-time construction noise would occur at low levels across the Sizewell Marshes and, particularly, Minsmere South Levels, as indicated by the predicted LA_{max} [impulsive noise] and LA_{eq} [chronic noise] values… Thus, noise levels would be below 55dB L_{Amax} and 45 L_{Aeq} across almost all of the Minsmere South Levels, and much of the Sizewell Marshes".

3.309 This means that some ecologically important areas will experience noise levels above these thresholds. We acknowledge that ES Vol. 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment[155] indicates that night-time noise could be most significant during Phases 3 and 4 due to the operation of the rail route and potentially night-time tunnelling activities. However, it is explained that freight unloading during Phase 2 may also produce significant night-time noise and we therefore query why the night-time noise modelling presented only covers these two phases and not Phases 1, 2 and 5.

3.310 We also query the scope of the updates to the noise modelling following the Changes to the Application and whether for example the modelling of night-time noise incorporates the additional freight trains proposed as part of the Change Application.

3.311 We also note our disagreement with the assertion in Table 2.34 of the Vol. 1 ES Addendum Ch. 2 Main Development Site[156] that the increased numbers of night time movements of freight trains will not affect birds to a different level to that originally assessed. The increased numbers of trains have the potential to increase chronic noise disturbance and/or lead to repeated disturbance (repeated instances of peak noise levels) and could therefore affect birds on the Minsmere South Levels and Sizewell Marshes.

Visual impacts

3.312 This discussion of visual disturbance is based on the maps presented in Figure 8.7 in the Shadow HRA Report Vol. 1 Part 3[157] and the updated map in Figure 8A.7 of the Shadow HRA Report Addendum Appendices 1A-10A Part 5[158] (although note that the extent of the visual buffer has not changed despite the Changes to the Application). From these it can be seen that for the majority of areas the visual buffer falls within areas also affected by noise disturbance, however, there are notable exceptions where the visual buffer indicates additional impacts, including the southern part of Minsmere South Levels, the central part of the eastern section of Sizewell Marshes and part of the western section of Sizewell Marshes.

3.313 It should also be noted that these maps only allow the comparison of noise and visual impacts for Phase 1 of construction when noise impacts are greatest. During phases with lower noise impacts larger areas of both Minsmere South Levels and Sizewell Marshes will be affected by visual disturbance alone. This has the effect of increasing the duration of significant impacts throughout the construction period when noise and visual disturbance are considered together.

Impacts on breeding birds of the Minsmere-Walberswick SPA

Impacts on breeding gadwall

3.314 Gadwall breeding on the Minsmere South Levels and Sizewell Marshes are acknowledged to be functionally linked to the Minsmere-Walberswick SPA population, which we welcome and

[155] ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204)
[156] Vol. 1 ES Addendum Ch. 2 Main Development Site (AS-181)
[157] Shadow HRA Report Vol. 1 Part 3 (APP-147)
[158] Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)

consider to align with the NE Guidance (and previous decisions) regarding functionally linked populations (Chapman and Tyldesley, 2016[159]).

3.315 Paragraph 8.8.360 of the Shadow HRA Report Vol. 1 Part 1[160] explains that, based on a wider SPA population of 106 pairs (including birds breeding on land within the SPA and on the Minsmere South Levels and Sizewell Marshes) these two areas hold 16% and 5% of the wider SPA population respectively.

3.316 The Shadow HRA Report goes on to assess the extent of encroachment of the 65dB L_{Amax} contour onto the Minsmere South Levels and Sizewell Marshes during Phase 1 of the construction period. This assessment is summarised and updated in sections 8.3 b) and f) of the Shadow HRA Report Addendum[161]. It is noted that distributional data for gadwall were only collected in 2020 and therefore the assessment of numbers of gadwall affected is based on an assumed even distribution across the South Levels and Sizewell Marshes. Given the encroachment of the 65dB L_{Amax} contour onto around 40% of the Minsmere South Levels and the majority of Sizewell Marshes during Phase 1 of construction is the Applicant has calculated that 11% of the wider SPA breeding gadwall population will be displaced due to impacts on the South Levels and Sizewell Marshes.

3.317 Paragraph 8.8.362 of the Shadow HRA Report goes on to state that this level of displacement would not apply to the whole construction period as the encroachment of the 65 dB L_{Amax} contour is lower in the subsequent construction phases and because the modelling for each phase is based on the worst case scenario within that phase and is not likely to occur for the full duration. However, we note our concerns above regarding overlapping construction phases, the lack of certainty regarding construction timelines and concerns that the impacts of construction Phase 5 have not been fully considered.

3.318 Paragraph 8.3.30 of the Shadow HRA Report Addendum also states that impacts are likely to be overestimated based on the distribution data obtained from the 2020 surveys. These indicated that the majority of breeding gadwall on the South Levels in that year were confined to pools in the north eastern corner of the levels which are not likely to be affected by significant noise levels or visual impacts. However, such generalisations should be treated with caution as the data are based on one year of surveys only, and as noted in the Shadow HRA Report, suitable habitat is found across the South Levels. Patterns of waterbird usage of the South Levels may vary year on year depending on water levels.

3.319 The Shadow HRA Report Addendum discusses the additional modelling of night-time noise and chronic noise. Whilst we have some concerns about the scope of these assessments, as noted above, the additional information is welcomed. With regard to chronic noise, paragraph 8.3.14 notes that the areas expected to be affected are those acknowledged to be affected by impulsive noise hence the conclusions of those original assessments are not changed. However, we note that chronic noise impacts on Minsmere South Levels and Sizewell Marshes do appear significant during Phases 3 and 4 of the construction period (and we assume in other phases) and therefore have the potential to exacerbate impacts on breeding birds in particular, as discussed above.

3.320 Paragraph 8.3.15 discusses the outputs of the night-time noise modelling and explains that "noise levels would be below 55dB L_{Amax} and 45 L_{Aeq} across almost all of the Minsmere South Levels, and much of the Sizewell Marshes." Based on Figure 8A.5 in the Shadow HRA Report Addendum

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159 Chapman, C. & Tyldesley, D. (2016) Functional linkage: How areas that are functionally linked to European sites have been considered when they may be affected by plans and projects - a review of authoritative decisions. Natural England Commissioned Reports, Number 207, see in particular pages 10-14

160 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

161 Shadow HRA Report Addendum (AS-173)

Appendices 1A-10A Part 5[162] significant parts of the eastern block of Sizewell Marshes would be affected by peak noise levels of above 55dB L_{Amax} overnight during Phases 3 and 4 along with the southern extremities of the Minsmere South Levels. Again, this lends weight to the need to consider the potential for significant effects throughout the construction period rather than just in Phase 1.

3.321 The construction of the flood compensation area and wetland habitat is stated not to affect breeding gadwall as this will occur in the first winter of Phase 1 of the construction period. As discussed above, we are concerned that should timelines slip and this construction extend into the following breeding season, impacts on breeding birds, including breeding gadwall, could be more significant than predicted.

3.322 The Shadow HRA Report and the Shadow HRA Report Addendum conclude that *adverse effect on* integrity arising from noise and visual impacts on breeding gadwall of the Minsmere-Walberswick SPA can be ruled out. In justifying this conclusion, paragraph 8.8.363 of the Shadow HRA Report states that the affected birds are part of the functionally linked population rather than the designated population and hence their displacement would not act to reduce the designated SPA population. We do not agree that any distinction can be drawn between designated and functionally linked populations for the purposes of HRA, since as agreed by the Applicant, these *linked* areas are equally important to these species. This statement appears to be inconsistent with the approach taken for marsh harrier within this Application and with the guidance regarding functionally linked populations produced by Natural England (Chapman and Tyldesley, 2016[163]), which states that:

> *"If effects on functionally linked land or sea are likely to have a significant effect on the population of species for which a European site was designated or classified, those effects must be considered fully in a Habitats Regulations Assessment."*

3.323 It is also noted that the current population is around three times higher than the population at the time of designation and again this is used to conclude that the impacts will not be significant, with paragraph 8.8.363 stating that:

> "*This* [displacement]… *would not prevent achievement of the supplementary advice on the generic conservation objectives to maintain the SPA population size of this qualifying feature at above 24 breeding pairs, whilst avoiding deterioration from its current level"*

3.324 The Supplementary Advice on Conservation Objectives[164] for breeding gadwall at this SPA sets a target to:

> *"Maintain the size of the breeding population at a level which is above 24 pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent."*

3.325 The Supplementary Advice on Conservation Objectives also note that the population has increased, with a five year peak mean count for the SPA of 84 breeding pairs between 2011/12 and 2015/16. We consider that displacement of 11% of the SPA population would constitute a significant deterioration from the current population level and would therefore compromise the

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[162] Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)

[163] Chapman, C. & Tyldesley, D. (2016) Functional linkage: How areas that are functionally linked to European sites have been considered when they may be affected by plans and projects - a review of authoritative decisions. Natural England Commissioned Reports, Number 207

[164] Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*. It does not appear to be possible download the entirety of the Supplementary Advice to provide page numbers

ability of the site to meet its conservation objectives. The level of displacement predicted, and our concerns outlined above mean we do not agree that *adverse effect on integrity* can be ruled out based on noise and visual impacts on breeding gadwall of the Minsmere-Walberswick SPA.

Impacts on breeding shoveler

3.326 Shoveler breeding on the Minsmere South Levels and Sizewell Marshes are acknowledged to be functionally linked to the Minsmere-Walberswick SPA population. Paragraph 8.8.384 of the Shadow HRA Report Vol. 1 Part 1[165] explains that, based on a wider SPA population of 77 pairs (including birds breeding on land within the SPA and on the Minsmere South Levels and Sizewell Marshes) these two areas hold 16% and 0.5% of the wider SPA population respectively. The Shadow HRA Report goes on to assess the extent of encroachment of the 65dB L_{Amax} contour onto the Minsmere South Levels and Sizewell Marshes during Phase 1 of the construction period. This assessment is summarised and updated in sections 8.3 b) and g) of the Shadow HRA Report Addendum[166]. It is noted that distributional data for shoveler were only collected in 2020 and therefore the assessment of numbers of shoveler affected is based on an assumed even distribution across the South Levels and Sizewell Marshes. Therefore given the encroachment of the 65dB L_{Amax} contour onto around 40% of the Minsmere South Levels and the majority of Sizewell Marshes during Phase 1 of construction it is calculated that 7% of the wider SPA breeding shoveler population will be displaced due to impacts on the South Levels and Sizewell Marshes.

3.327 The discussion of potential impacts on breeding shoveler in the Shadow HRA Report and Shadow HRA Report Addendum covers the same points as those covered for breeding gadwall above hence, we will not repeat our concerns in full, but our key points can be summarised as follows:

- Concerns regarding overlapping construction phases, the lack of certainty regarding construction timelines and that the impacts of construction Phase 5 have not been fully considered
- Conclusions regarding shoveler distribution should be treated with caution as the data are based on one year of surveys only and suitable habitat is found across the South Levels
- The assessments of night-time noise and chronic daytime noise during Phases 3 and 4 of construction indicate that significant effects are likely throughout the construction period rather than just in Phase 1.
- Should timelines for the construction of the flood compensation area and wetland in the first winter of Phase 1 of construction slip and this construction extend into the following breeding season, impacts on breeding shoveler could be more significant than predicted
- The significance of impacts should not be disregarded or downgraded because they affect a functionally linked population
- Whilst the current population level may be above that at the time of designation, any deterioration from current population levels would compromise the site's ability to meet the requirement to "avoid deterioration from its current level" set in the Supplementary Advice on Conservation Objectives[167].

3.328 Therefore, we do not agree that *adverse effect on integrity* on the breeding shoveler population of the Minsmere-Walberswick SPA can be ruled out.

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165 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

166 Shadow HRA Report Addendum (AS-173)

167 Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features.*

Impacts on breeding teal

3.329 Impacts on breeding teal are ruled out on the basis of sporadic recent breeding on the Minsmere South Levels and Sizewell Marshes and because noise impacts are not expected on the main areas of habitat suitable for breeding teal. In our view, this is not accurate as much of the South Levels are likely to provide suitable habitat.

3.330 It is notable that evidence of potential breeding (1 pair) was recorded during the 2020 surveys on the South Levels, although breeding is difficult to ascertain unless young are seen. Natural England's Supplementary Advice on the Conservation Objectives[168] for the site note that the population of breeding teal at the time of the SPA's designation was 73 pairs, but since that point the population has declined dramatically to a five year mean peak count (2011/12 - 2015/16) of one pair. The conservation objectives for the site therefore require the feature to be restored, although it is noted that the reasons for the decline are not understood.

3.331 Therefore, the potential for the proposed development to affect the ability of conservation measures to restore this feature (if causes are understood and suitable solutions can be proposed) should be considered.

Impacts on breeding bittern

3.332 Construction of the flood compensation area and wetland is planned for the first winter of construction Phase 1. We are concerned that, if this work were to stretch into the breeding period, the 65dB$_{LAmax}$ contour stretches into the Minsmere North Levels reedbeds (part of the SPA) and has potential for impacts on breeding bittern.

3.333 It should be noted that bitterns typically start booming[169] in February and that noise disturbance could have the potential to interfere with this. We therefore recommend that the commitment to carry out this work in the winter period (October – February inclusive) must be secured through a condition to ensure that this mitigation can be relied on and to justify conclusions.

Impacts on breeding marsh harrier

3.334 Noise and visual impacts on breeding marsh harrier of the Minsmere-Walberswick SPA are acknowledged to result in *adverse effect on integrity* and are discussed in detail below.

Impacts on non-breeding birds of the Minsmere-Walberswick SPA

Impacts on non-breeding gadwall

3.335 Paragraphs 8.8.469 – 492 of the Shadow HRA Report Vol. 1 Part 1[170] discuss impacts on non-breeding gadwall of the Minsmere-Walberswick SPA. It is acknowledged that the Minsmere South Levels hold an average of 53% of the SPA population (range 5-70%) and Sizewell Marshes hold 4-18% and that these populations are functionally linked to the SPA. It is noted that a high proportion of the gadwall which use Sizewell Marshes may be affected by construction-related disturbance although impacts on the South Levels are considered unlikely as the areas used by gadwall are outside the noise and visual disturbance zones, with the main area used by wintering gadwall on the South Levels likely to experience peak noise levels of approximately 64dB L_{Amax} during Phase 1 of construction and of between 54 and 64dB L_{Amax} during Phases 2 to 5. No

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168 Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

169 The breeding call of male bitterns

170 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

estimates of the numbers of birds or proportion of the SPA population likely to be displaced are presented. This omission makes it difficult to assess the likely significance of the impacts.

3.336 The Shadow HRA Report makes reference to a small number of studies that indicate gadwall are likely to make regular movements between sites during winter. These are used to conclude that impact on Sizewell Marshes (and therefore on the SPA population) are overestimated. As this discussion refers to non-breeding birds, we consider that any such movements are a part of the natural function of the SPA population and that the key point should be whether the SPA (and functionally linked areas) are able to continue to support current numbers of birds. If noise and/or visual disturbance mean that part of the supporting habitat is unable to provide that function and that the numbers of birds the SPA is capable of supporting reduces as a result, we consider this an indication of impact.

3.337 It is stated in paragraph 8.8.490 of the Shadow HRA Report that impacts on the ability to achieve the site's conservation objectives are unlikely given that the population is currently at more than three times the level at the time of designation. The Supplementary Advice on Conservation Objectives[171] for non-breeding gadwall at this SPA sets a target to:

> *"Maintain the size of the non-breeding population at a level which is above 90 individuals, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent."*

3.338 The Supplementary Advice on Conservation Objectives also note that the population has increased, with a five year peak mean count for the SPA of 440 individuals between 2012/13 and 2016/17. We consider that displacement of a likely minimum of 4% of the SPA population (based on the lowest estimates above of the total population of Sizewell Marshes and noting that a displacement figure is not presented) would constitute a significant deterioration from the current population level and would therefore compromise the ability of the site to meet its objectives. The level of displacement predicted, and our concerns outlined above mean we do not agree that *adverse effect on integrity* can be ruled out based on noise and visual impacts on non-breeding gadwall of the Minsmere-Walberswick SPA.

Impacts on non-breeding shoveler

3.339 Paragraphs 8.8.510 – 527 of the Shadow HRA Report Vol. 1 Part 1[172] discuss impacts on non-breeding shoveler of the Minsmere-Walberswick SPA. It is acknowledged that the Minsmere South Levels hold an average of 49% of the SPA population (range 16-66%) and Sizewell Marshes hold 4-10% and that these populations are functionally linked to the SPA. It is noted that any shoveler which use Sizewell Marshes may be affected by construction-related disturbance although impacts on the South Levels are considered unlikely as the main areas used by shoveler are outside the noise and visual disturbance zones, with the main area used by wintering shoveler on the South Levels likely to experience peak noise levels of approximately 64dB L_{Amax} during Phase 1 of construction and of between 54 and 64dB L_{Amax} during Phases 2 to 5. As for gadwall above, no estimates of the numbers of birds or proportion of the SPA population likely to be displaced are presented. This omission makes it difficult to assess the likely significance of the impacts.

3.340 The discussion of potential impacts on non-breeding shoveler in the Shadow HRA Report and Shadow HRA Report Addendum covers the same points as those covered for non-breeding

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171 Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

172 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

gadwall above hence, we will not repeat our concerns in full, but our key points can be summarised as follows:

- Movements of birds within and between sites are a part of the natural function of the SPA population and that the key point should be that if noise and/or visual disturbance mean that the ability of part of the SPA or functionally linked areas to continue to support current numbers of birds is reduced, this is an indication of impact.
- Whilst the current population level may be above that at the time of designation, any deterioration from current population levels would compromise the site's ability to meet the requirement to "avoid deterioration from its current level" set in the Supplementary Advice on Conservation Objectives[173].

3.341 The level of displacement predicted, and our concerns outlined above mean we do not agree that *adverse effect on integrity* can be ruled out based on noise and visual impacts on non-breeding shoveler of the Minsmere-Walberswick SPA.

Impacts on non-breeding white-fronted goose

3.342 Paragraphs 8.8.541 - 542 of the Shadow HRA Vol. 1 Part 1[174] discusses potential noise impacts on non-breeding white-fronted geese which roost on the Minsmere South Levels. It is stated that the closest areas of open water which are likely to be used as roosts occur beyond the 64dB L_{Amax} contour for daytime noise levels during all construction phases, so that disturbance from noise impacts to the roosting birds is unlikely and that as roosting birds will only be present at night, noise levels will be "considerably lower". However, given that daytime, for the purposes of noise modelling, is defined as 7:00 AM to 11:00 PM and that these roosts are present in winter, roosting white fronted geese are likely to be exposed to daytime noise levels during the hours of darkness. We also note that, to date, surveys of nocturnal white-fronted goose usage of the South Levels have not been presented so it is not possible to attach any confidence to conclusions regarding the likely noise levels experienced by roosting geese.

3.343 However, we note that the Shadow HRA Report Addendum[175] states that these surveys have been carried out over winter 2020-2021 and we look forward to seeing the outputs of these surveys.

3.344 White-fronted geese commute from their main feeding areas at North Warren and the roosting sites at Minsmere around 2-3 hours after sunset and, as acknowledged in paragraph 8.8.541 of the Shadow HRA Report, a direct flight would take birds over the main development site. The Shadow HRA Report Addendum in paragraph 8.3.55 concludes that the resulting disturbance to commuting birds is:

> *"likely to result in small deviations to the flight routes only, and are considered highly unlikely to prevent birds from using the roosting areas".*

3.345 Given that early sunsets in winter mean that these flights are likely to be exposed to daytime construction noise levels, we consider that impacts could be more significant than assessed.

Impacts on birds of the Minsmere-Walberswick Ramsar site

3.346 Section 8.9 of the Shadow HRA Report Vol. 1 Part 1[176] covers impacts on the Minsmere-Walberswick Ramsar site and states that as the breeding assemblage species designated under

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[173] Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

[174] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

[175] Shadow HRA Report Addendum (AS-173)

[176] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

Ramsar criterion 2 are also SPA features (with the exception of bearded tit, which is briefly discussed in paragraph 8.9.6), the assessment of impacts is covered by the SPA section. Therefore, our comments above under the Minsmere-Walberswick SPA section for breeding bittern, gadwall, teal and shoveler also apply to the Minsmere-Walberswick Ramsar site.

Impacts on breeding birds of the Sandlings SPA

Woodlark and nightjar

3.347 Paragraph 14.12.106 of ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[177] explains that:

> *"The north-west extremity of a relatively small, discrete, block of the [Sandlings] SPA approaches to within 0.7 km to the south of the main development site, with this discrete block likely to support approximately 3% of the SPA breeding nightjar population and approximately 9% of the SPA breeding woodlark population."*

3.348 It is stated that this area is within the zone of visual impacts although outside the area of predicted significant noise impacts. The Shadow HRA Report concludes that no *adverse effect on integrity* will arise as a result, but no justification is given for this conclusion. Given the significant percentage of the SPA population affected for both these species, we consider that potential impacts warrant further assessment.

3.349 Paragraphs 3.7.6-7 of the Shadow HRA Report Addendum Appendix 1A Further Analysis of Inter-Pathway Effects[178] also consider that the proportion of the SPA populations affected are small and there is no potential for inter pathway effects to occur when combined with other impacts from the Sizewell C proposals. This assessment therefore fails to consider the potential for combined effects of recreational pressure and visual disturbance during construction on this part of the Sandlings SPA and we therefore consider the assessment requires updating.

Impacts on the Minsmere-Walberswick Heaths and Marshes SSSI and Sizewell Marshes SSSI

3.350 Paragraph 14.12.117 of ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[179] discusses the potential impacts of noise and visual disturbance on the Minsmere-Walberswick Heaths and Marshes SSSI and explains that during Phase 1 of construction the encroachment of the 70dB L_{Amax} contour covers:

> *"…approximately 16.4ha (0.7% of the total 2,325.89ha) of the Minsmere to Walberswick Heaths and Marshes SSSI. During Phase 2, this reduces to 13.4ha (0.53%) with virtually no encroachment of the 70dB L_{Amax} contour during Phases 3, 4 and 5…"*

3.351 No assessment is made of feature species[180] found in the area most affected or their sensitivity to impacts. Also, no discussion of the encroachment of the 65dB L_{Amax} contour relevant to breeding birds is provided. We also note that as the SSSI underpins the Minsmere-Walberswick SPA our conclusions relating to the SPA are also relevant here.

3.352 With regard to the Sizewell Marshes SSSI, paragraph 14.12.121 of the same document states that:

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177 ES Vol 2 Chapter 14 Terrestrial Ecology and Ornithology (AS-033)
178 Shadow HRA Report Addendum Appendix 1A Further Analysis of Inter-Pathway Effects (AS-174)
179 ES Vol 2 Chapter 14 Terrestrial Ecology and Ornithology (AS-033)
180 Species relevant to this assessment could include (but not necessarily be limited to) bittern, marsh harrier, snipe, lapwing, gadwall, shoveler and teal.

"…the noise contour modelling figures show that during Phases 1 and 2 of the construction the 70 dB L_{Amax} contour would encroach across approximately 57ha of Sizewell Marshes SSSI (55% of Sizewell Marshes SSSI total of 194.33ha). During Phases 3 and 4 this would reduce to approximately 14ha (13% of Sizewell Marshes SSSI). The 65dB L_{Amax} contour would encroach further in both cases"

3.353 This indicates that impacts of noise on Sizewell Marshes SSSI are likely to be significant. Paragraph 14.12.125 goes on to state that:

"Sizewell Marshes SSSI supports a diverse bird assemblage but populations are relatively small … Potential disturbance impacts are considered to be a short-term but of large magnitude. For any given species the impacts on a relatively small number of individuals associated with the SSSI would not result in a significant effect on any of the overall populations for any species."

3.354 Again, no assessment has been made of the feature species likely to be affected or their sensitivity to impacts and the conclusion around overall bird populations does not give adequate recognition to the importance or integrity of the SSSI itself. We therefore cannot agree with the conclusions of the assessment as provided.

Monitoring

3.355 Given the potential impacts of noise and visual disturbance on both the Minsmere South Levels and Sizewell Marshes, we consider that a robust programme of monitoring should be proposed for the construction period. A monitoring programme should include both monitoring of noise levels for the purpose of verification of noise modelling predictions and monitoring of the distribution of key bird species to detect impacts and this programme should be secured through a requirement in the DCO with the need for additional mitigation to be considered should noise levels and/or potential impacts be higher than predicted.

In-combination impacts

Offshore windfarm cable routes

3.356 Vol. 3 Ch. 10 Inter-relationships, Project-wide, Cumulative and Transboundary Effects Appendix C[181], provides an assessment of cumulative terrestrial ecology and ornithology effects with the onshore cable route construction of the Scottish Power Renewables (SPR) offshore wind schemes. This notes the potential for in-combination effects of direct land take and disturbance from the SPR projects along with recreational pressure and noise and visual disturbance from Sizewell C construction.

3.357 It concludes that since the main part of the SPA to be directly affected, the 0.7km SPA block "only" supports approximately 3% of the SPA breeding nightjar population and approximately 9% of the SPA breeding woodlark population that, with the mitigation proposed by both developers, effects would not be significant. This assessment does not consider the potential for displacement due to the construction of this Application to increase/change the predicted impacts of the SPR projects and we therefore consider that this assessment requires updating.

[181] Vol. 3 Ch. 10 Inter-relationships, Project-wide, Cumulative and Transboundary Effects Appendix C (AS-201)

Sizewell B relocated facilities works

3.358 We note that paragraph 1.1.4 of ES Vol. 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment[182] explains that:

> *"the noise impact assessment for the relocation of Sizewell B facilities was undertaken as a separate exercise, and noise sources and activities associated with relocated facilities were therefore not included in the prediction models".*

3.359 Given that the relocated facilities work is expected to take place over a period of 4.5 years and overlap with Sizewell C construction Phase 1 and part of Phase 2, the impacts of both works must be considered together and combined noise modelling outputs should be presented.

Conclusions regarding noise and visual impacts on birds

3.360 In conclusion, based on the above concerns and the information available at present, we do not agree that *adverse effect on integrity* can be ruled out for the following sites and species for impacts from the project alone:

- Breeding gadwall and shoveler and non-breeding gadwall, shoveler and white-fronted goose of the Minsmere-Walberswick SPA and Ramsar site

3.361 We consider that the Applicant must consider how the mitigation hierarchy could be applied to avoid, reduce, mitigate or, where appropriate, compensate this impact.

3.362 We also consider that further information is required to support the conclusions for breeding bittern and teal of this site.

3.363 We agree with the conclusion that *adverse effect on integrity* cannot be ruled out for breeding marsh harrier of the Minsmere-Walberswick SPA and Ramsar site and this is discussed further below.

3.364 We also consider that significant impacts from the Application's alone are likely on the following sites:

- Minsmere-Walberswick Heaths and Marshes SSSI
- Sizewell Marshes SSSI

3.365 In relation to this project in combination with other projects, we do not agree that *adverse effect on integrity* can be ruled out for the following site and species:

- Breeding woodlark and nightjar of the Sandlings SPA

Noise and visual disturbance to marsh harriers of the Minsmere-Walberswick SPA and Ramsar site

Background

3.366 The following sections discuss the assessment of noise and visual disturbance effects on marsh harrier at RSPB Minsmere (part of the Minsmere-Walberswick SPA and Ramsar site). Our comments relate primarily to the assessment of impacts and conclusions regarding AEOI in the

[182] ES Vol 2 Ch. 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204)

Shadow HRA Report Vol. 1 Part 1[183] and the proposals for compensatory habitat set out in the Shadow HRA Report Vol. 4: Compensatory Measures[184].

3.367 In brief, it has been concluded that noise and visual disturbance is likely to result in significant displacement of breeding marsh harrier from their functionally linked foraging areas on the Minsmere South Levels and Sizewell Marshes. This loss of foraging habitat is proposed to be compensated through the provision of habitats managed to provide an uplift in marsh harrier prey abundance and availability. However, we have significant concerns regarding the sufficiency of the compensation proposed. Note that marsh harriers are features of both the SPA and Ramsar site and therefore although in our comments below we refer to the SPA, these comments also apply to the Ramsar site.

The Minsmere-Walberswick SPA and Ramsar site marsh harrier population

Population at designation

3.368 At the time of SPA designation in 1991, the Minsmere-Walberswick SPA[185] supported 15 breeding females which represented 20% of the British population.

Site Conservation Objectives and Supplementary Advice

3.369 The SPA site conservation objectives[186] relevant to the breeding marsh harrier population are to:

> *"Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;*
>
> - *The extent and distribution of the habitats of the qualifying features*
> - *The structure and function of the habitats of the qualifying features*
> - *The supporting processes on which the habitats of the qualifying features rely*
> - *The population of each of the qualifying features, and,*
> - *The distribution of the qualifying features within the site."*

3.370 Natural England's Supplementary Advice on Conservation Objectives[187] specific to the marsh harrier feature sets a target to:

> *"Maintain the size of the breeding population at a level which is above 15 breeding females, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent."*

3.371 It is noted that the marsh harrier population is dynamic and that this target represents a minimum value for conservation measures to achieve. Importantly, it is also stated that:

> *"Given the likely fluctuations in numbers over time, any impact assessments should focus on the current size of the site's population, as derived from the latest known or estimated level established using the best available data. This advice accords with the obligation to avoid deterioration of the site or significant disturbance of the species for which the site is classified,*

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183 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
184 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)
185 The SPA designation - http://publications.naturalengland.org.uk/file/5944954236960768
186 The SPA's generic Conservation Objectives - http://publications.naturalengland.org.uk/file/4811128974868480
187 Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features.*

and seeks to avoid plans or projects that may affect the site giving rise to the risk of deterioration."

3.372 A number of targets are also set relating to supporting habitats which are relevant to the issues raised by this Application. These include the following targets:

- Restrict the frequency, duration and / or intensity of disturbance affecting roosting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed – it is noted that such effects may operate within or outside the site boundary and include changes to feeding behaviour.
- Maintain the extent, distribution and availability of suitable habitat (either within or outside the site boundary) which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding) – it is noted that the target applies year-round, but that quantified figures for extent are not available.
- Maintain the distribution, abundance and availability of key food and prey items (e.g. mammals, birds) at preferred sizes (e.g. voles, mice, rabbit; birds of pipit to duck size) – it is noted that direct or indirect impacts which may affect the distribution, abundance and availability of prey may adversely affect the population and alter the distribution of birds.
- Maintain safe passage of birds moving between nesting and feeding areas – this is noted as critical to their breeding success and to the adult fitness and survival and the target applies within the site boundary and where birds regularly move to and from off-site habitat.

3.373 Note that these targets are set in relation to the SPA but would of course equally benefit the Ramsar site.

Current SPA and Ramsar site population and trends

3.374 The SPA population remains slightly above the Supplementary Advice on Conservation Objectives[188] target of a minimum of 15 pairs, but has experienced a decline since the early 2000s, as can be seen in Table 1 below. The causes of this are not currently clear, and the Site Improvement Plan notes a need for further research in this area.

[188] Natural England Supplementary Advice on Conservation Objectives (SACOs) - Minsmere-Walberswick SPA, please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

Table 1 Marsh harrier nests within the Minsmere-Walberswick SPA and Ramsar site 2004-2020 (updated from Table 6.6 in the Shadow HRA Report Vol. 1 Part 1[189])

Year	Minsmere	Hen Reedbed	Walberswick NNR	Total
2004	10	2	9	21
2005	12	4	8	24
2006	12*	4	7	23
2007	17	6	8	31
2008	15	3	no data	At least 18
2009	12	3	9	24
2010	10	3	no data	At least 13
2011	12	3	7	22
2012	10	3	8	21
2013	9	2	no data	At least 11
2014	6	2	5	13
2015	9	2	no data	At least 11
2016	9	2	6	17
2017	9	2	6	17
2018	8	3	6	17
2019	10	3	no data	At least 13
2020	no data	2	no data	insufficient data

*Table 6.6. in APP-145 incorrectly reports 13 nests in 2006

3.375 Table 2 below shows the breeding productivity (average fledged young per nest) at RSPB Minsmere in the last 10 years. It can be seen that the decline noted across the SPA is apparent at Minsmere, with a five year mean of 13.2 nests for 2005 to 2009, and a five year mean of 9 from 2015 to 2019 and a possible slight decline in productivity, although this is less clear.

Table 2 Breeding marsh harrier productivity at RSPB Minsmere (part of the Minsmere-Walberswick SPA and Ramsar site) since 2005

Year	Nests at RSPB Minsmere	Fledged young (productivity)
2005	12	34 (2.83)
2006	12	27 (2.25)
2007	17	38 (2.24)
2008	15	23 (1.53)
2009	12	23 (1.92)
2010	10	17 (1.70)
2011	12	21 (1.75)
2012	10	24 (2.40)
2013	9	15 (1.67)
2014	6	10 (1.67)
2015	9	14 (1.56)
2016	9	18 (2.00)
2017	9	19 (2.11)
2018	8	12 (1.50)
2019	10	12 (1.20)
2020*	no data	no data

*Full surveys were not carried out in 2020 due to covid-19

[189] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

Marsh harrier sensitivity to noise and visual disturbance

3.376 Our comments below relate to the evidence base for the assessment of the sensitivity of marsh harriers to noise and visual disturbance impacts.

Threshold for noise disturbance

3.377 We note the statement in paragraph 8.8.192 of the Shadow HRA Report Vol. 1 Part 1[190] that a 70dB L_{Amax} threshold has been proposed to represent the level at which a potentially adverse behavioural response could occur.

3.378 We are concerned about specific effects of noise on foraging marsh harriers. It is unclear how increased noise levels will affect the behavioural response of foraging birds, as the effect most likely to be in question is avoidance of the area rather than flushing. As avoidance may be less energetically costly than flushing (and foraging birds are less constrained to a particular location than when at the nest), it may be that birds will display avoidance at lower noise levels than those which would flush birds already present in an area.

3.379 As discussed in paragraph 8.8.175, it is also known that harriers in general make significant use of their hearing when foraging, to enable them to detect prey, particularly within taller vegetation. Marsh harriers (as in other harrier species) have large ear openings relative to their size, and a facial disc similar in structure to those of owl species. Rice (1982) (cited in Clarke, 1995[191]) found that the directional hearing of northern harriers is similar to that of owl species.

3.380 Whilst marsh harriers may employ hearing to a slightly lesser degree than other harriers when hunting (due to differences in foraging flight height), it is thought that hearing may be as important as sight for foraging marsh harriers (Clarke, 1995). It is therefore likely that foraging harriers may be less efficient if construction noise levels mask the auditory cues which are important for prey detection, and that this may result in avoidance of affected areas and/or reduced efficiency of prey capture.

3.381 Despite the concerns above, we acknowledge that there is a very limited evidence base with which to generate a species-specific noise disturbance threshold for marsh harrier. Therefore, we have concluded that, based on currently available information, there is insufficient *alternative* evidence to challenge the 70dB L_{Amax} threshold set by the Applicant. However, concerns remain, and it is therefore important that the proposed compensation is adequate in terms of area, location, and the success of measures proposed to increase prey availability.

Marsh harrier sensitivity to visual disturbance from human activity

3.382 Paragraphs 8.8.207 – 8.8.208 of the Shadow HRA Report Vol. 1 Part 1[192] quotes Madders and Whitfield (2006)[193] regarding the sensitivity of marsh harrier to windfarms. This study reports that the sensitivity of marsh harriers to displacement by operational windfarms is "Low?". The question mark indicates that not enough studies were available to draw a definite conclusion for the species, indeed, in the case of marsh harrier, the conclusion is based on two pers. comm. references rather than published studies.

3.383 It should also be queried as to how relevant the effects of an operational windfarm are compared to a large-scale construction project. Madders and Whitfield (2006) makes no further reference

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190 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
191 Clarke, R. (1995) The Marsh Harrier. Hamlyn Species Guides. London
192 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
193 Madders, M. & Whitfield, D.P (2006) Upland raptors and the assessment of wind farm impacts. Ibis 148: 43-56

to marsh harriers, perhaps as this paper focuses on upland areas. For comparison, Clarke (1995)[194] reports studies showing that marsh harriers avoid nesting, or even flying, near areas used by people and Alves *et al.* (2014)[195] studying habitat preferences found that marsh harriers using arable areas showed clear disturbance and avoidance when farmers and machines were present. They noted that:

> *"Due to high human pressure, individuals may be forced to seek less favourable locations, which may ultimately have a negative impact in reproductive success and thus lead to a population decrease."* (Alves *et al.* 2014)

3.384 It is therefore unclear why, when the paragraphs 8.8.213 – 215 of the Shadow HRA Report Vol. 1 Part 1 quotes Ruddock and Whitfield (2007)[196] who recommend that a 300-500m buffer be applied for visual disturbance to marsh harrier at their breeding locations, a buffer of only 150m for visual disturbance has been applied by the Applicant[197]. Whilst we acknowledge that the buffer for this development applies to foraging birds rather than birds at the nest, no evidence is presented to support the conclusion.

Assessment of baseline conditions – current noise levels and habitat usage by marsh harriers

3.385 The points below relate to the discussion within the Applicant's assessments of baseline conditions with regard existing noise levels and habitat use relevant to marsh harriers at Minsmere.

Comparison of predicted noise levels with those recorded at Minsmere

3.386 As discussed above regarding noise impacts on breeding birds in general, the Shadow HRA Report Vol. 1 Part 1[198] paragraphs 8.8.185-186 refers to a survey of noise levels at RSPB Minsmere conducted near South Hide which is used to assess noise levels experienced by breeding marsh harriers. The noise effects described include anthropogenic sources including motor vehicles and visitors, resulting in maximum sound levels of over 70dB. We note the acknowledgement that the acoustic data collected at Minsmere were from an area close to visitor routes and may not be representative of the wider area. South Hide is on a visitor route between the main scrape and the eastern end of the reedbed. Marsh harriers do not nest in close proximity to this route. This survey location is therefore not appropriate for assessing the noise levels likely to be experienced by breeding marsh harriers – it would not be expected that visitors and motor vehicles would result in the noise levels quoted 'at the bird' given the distance of the birds from these sources. Decay rates[199] would indicate that noise levels of 70-80dB at South Hide would have reduced significantly at the distances that nests are likely to be from these sources.

3.387 The location of the South Hide survey is also not within a primary foraging area for marsh harriers. The reduced foraging activity in this area could be influenced by a number of factors. These include the presence of the visitor trail and the close proximity to the main coastal lagoons (the

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194 Clarke, R. (1995) The Marsh Harrier. Hamlyn Species Guides. London

195 Alves, M., Ferreira, J., Torres, I., Fonseca, C. & Matos, M. (2014) Habitat Use and Selection of the Marsh Harrier *Circus aeruginosus* in an Agricultural-Wetland Mosaic. Ardeola. 61(2): 351-366

196 Ruddock, M. & Whitfield, D. P. (2007) A review of disturbance distances in selected bird species, Natural Research (Projects) for SNH

197 Use of the word "buffer" is perhaps misleading since it is used to describe the area in which impacts are assumed to occur so Ruddock and Whitfield (2007) are actually saying that *impacts would occur* within 300- 500 metres whereas the Applicant is saying impacts would only extend to 150 metres. At 150 metres this means that the area assumed to be affected by visual disturbance is within the area also assumed to be affected by noise disturbance.

198 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

199 The manner by which sound decreases with distance from the source

Scrape) where a high density of colonial nesting avocets, gulls and terns will attack marsh harriers and drive them away. Therefore, the impacts of noise alone are difficult to interpret at this location. It should also be clarified as to whether any atypical, noise producing, management work was being carried out at that location during the survey period.

3.388 With regard sound levels attributed to insect noise and birdsong, it is important to draw a distinction between anthropogenic noise which is likely to cause alarm to birds and noise sources that are natural (e.g. birdsong, the sea) and hence much less likely to cause disturbance.

Accuracy of flight location records and use of flight length vs flight number metrics

3.389 Paragraphs 6.6.76 – 6.3.84 of the Shadow HRA Report Vol. 1 Part 1[200] describe the flight path length and flight path number metrics used to analyse intensity of use of Minsmere South Levels and Sizewell Marshes by foraging marsh harriers. In general, we agree that the metric based on flight length is the more useful measure of marsh harrier activity. However, we note that that there may have been differences in recorded locations of flights depending on the position of the surveyor. While both the flight length and flight number metrics could be affected by any such bias, the flight length metric does assume a greater level of accuracy in recording and hence is potentially vulnerable to error.

Marsh harrier usage of Minsmere South Levels

3.390 We note the conclusion in paragraph 6.3.92 of the Shadow HRA Report Vol. 1 Part 1[201] that most flight activity is concentrated over the north western part of the South Levels. We agree that this is likely to be due to proximity to nesting areas, however, we also note that the majority of the South Levels are well used (R. Harvey[202], pers. comm.), and that use may vary depending on season and amount of surface water. For example, when the eastern parts of the Levels are flooded, they are likely to be productive for marsh harriers hunting waterfowl (A. Rowlands[203] pers. comm.).

Conclusions regarding noise and visual impacts on marsh harriers of the Minsmere-Walberswick SPA and Ramsar site

3.391 Our general comments on sources of noise and visual impacts during the construction period and the adequacy of the modelling of noise levels during the different phases of construction can be above in our overarching comments on noise and visual disturbance. The comments below are points which relate specifically to potential impacts on marsh harrier.

Noise disturbance of marsh harriers foraging over the Minsmere South Levels and Sizewell Marshes

3.392 Paragraphs 8.8.190 – 8.8.202 of the Shadow HRA Report Vol. 1 Part 1[204] discuss the extent of noise and visual disturbance predicted to affect the Minsmere South Levels and Sizewell Marshes during the construction phase, based on the outputs of the noise modelling shown in Figures 8.2 to 8.7 in Shadow HRA Report Vol. 1 Part 3[205]. It should be noted that both these areas are acknowledged to provide functionally linked foraging habitat for the SPA marsh harrier population.

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200 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
201 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
202 RSPB Minsmere Site Manager
203 RSPB Suffolk Area Manager
204 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
205 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 3 (APP-147)

3.393 Paragraphs 8.8.194 and 8.8.196 acknowledge that the 70dB L_{Amax} displacement threshold will be exceeded over the southernmost part of Minsmere South Levels and the eastern side and southern end of Sizewell Marshes during Phase 1 and Phase 2 of the construction period (see also Figures 8.3 and 8.4 of Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 3), with the extent predicted to be smaller in Phases 3 and 4 and greater again in Phase 5 (although slightly smaller than in Phases 1 and 2, and please note our concerns raised above regarding the mapped outputs for this stage). Paragraph 8.8.195 explains that these predictions are based on the WCS noise levels for each phase and that modelling across narrower time periods shows that these maximum levels will not occur throughout each phase. Paragraph 8.8.201 indicates that this assessment based on WCS noise levels is therefore highly precautionary, however, we note that it is standard practice to base such assessments on the WCS, that the modelling based on narrower time periods relies on the construction timetable proceeding exactly according to plan without delays or changes, which is unlikely to be the case, and that there are several sources of uncertainty in the assessment requiring a high level of precaution to be used, such as the uncertainty about the application of the 70dB L_{Amax} threshold to marsh harrier.

Acoustic fence

3.394 Paragraph 8.8.67 of the Shadow HRA Report Vol. 1 Part 1[206] explains that the impact assessment for the Phase 1 construction period is based on the assumption that a 5m high acoustic fence (or equivalent mitigation) will be in place. It is crucial that the 5m acoustic fence is constructed at the beginning of the Phase 1 construction period, especially as marsh harrier compensation is not likely to be fully functional at this point (we discuss this further below) and Phase 1 of the construction period is one of the phases with the highest noise impacts on this area.

Barrier effects on Sizewell Marshes

3.395 The baseline surveys of marsh harrier flight activity demonstrated that marsh harriers nesting at RSPB Minsmere not only forage over the wetland habitats of the Minsmere South Levels and Sizewell Marshes (as mentioned above), but also nearby arable habitats. Given the location of the main construction area in an arc between the nesting sites at RSPB Minsmere and the foraging areas at Sizewell Marshes and the significant human activity that will take place within this area (to which marsh harriers are known to be sensitive, as discussed above), we consider it unlikely that marsh harriers will continue to over-fly this area during the construction phase in order to access foraging habitats at Sizewell Marshes[207]. We therefore support the conclusion in paragraph 8.8.200 of the Shadow HRA Report Vol. 1 Part 1[208] that noise levels over the main development site (as well as visual disturbance) should be considered to lead to a "barrier effect" on foraging marsh harriers, meaning their complete displacement should be assumed from the entirety of Sizewell Marshes.

3.396 With regard to the discussion of marsh harrier use of Trimley Marshes (next to the Port of Felixstowe) in paragraphs 8.8.180-103 of the Shadow HRA Report Vol. 1 Part 1, it should be noted that conclusions about barrier effects cannot be drawn from this site. At Trimley, there is no suitable marsh harrier habitat in any location which would require harriers to fly over the Port in order to access it from their breeding sites, therefore this site provides no evidence from which inferences can be made about whether marsh harriers from Minsmere will continue to use Sizewell Marshes during the construction period.

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206 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

207 See Figures 6.2 – 6.7, pages 3–9, the Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 2 (APP-146) for marsh harrier flight activity and the red line boundary

208 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

Visual disturbance of marsh harriers foraging over the Minsmere South Levels and Sizewell Marshes

3.397 As discussed in the section on marsh harrier sensitivity to visual disturbance from human activity above, we have concerns about the Applicant's use of a 150m buffer[209] to represent the area around the main development site subject to visual disturbance displacement. Paragraphs 8.8.204 – 8.8.210 of the Shadow HRA Report Vol. 1 Part 1[210] discuss the application of this buffer, which is illustrated in Figure 8.10 in the Shadow HRA Report Vol. 1 Part 4[211]. Based on this buffer, visual displacement will affect the southern extremities of the Minsmere South Levels along with parts of the periphery of Sizewell Marshes, however, the visual buffer is generally encompassed within the area affected by the 70dB L_{Amax} noise displacement threshold. As discussed above, we consider that insufficient evidence has been presented to support deviation from the recommended 300-500m buffer (based on Ruddock and Whitfield, 2007)[212] and hence argue that this assessment risks underestimating impacts.

Conclusion that adverse effects on integrity on the Minsmere-Walberswick SPA and Ramsar site cannot be excluded

3.398 Due to the predicted extent of disturbance displacement affecting the Minsmere South Levels (based on the application of the 70dB L_{Amax} threshold and the visual buffer) and the predicted barrier effect resulting in complete displacement of foraging harriers from Sizewell Marshes SSSI during the construction period, we agree with the conclusion in paragraph 8.8.243 of the Shadow HRA Report Vol. 1 Part 1[213] that *adverse effect on integrity* on the Minsmere-Walberswick SPA (and Ramsar site) cannot be excluded. We therefore agree that, subject to consideration of *alternative solutions* and *imperative reasons of overriding public interest*, suitable compensation measures must be proposed.

Calculation of the compensatory requirement

Compensatory requirement based on area of habitats "lost"

3.399 As stated above, we agree with the Applicant's conclusion that the likelihood of barrier effects from the construction area will lead to complete displacement of marsh harrier from suitable foraging habitats within Sizewell Marshes. Paragraph 8.8.232 of the Shadow HRA Report Vol. 1 Part 1[214] explains that this loss of 78ha of wetland foraging from Sizewell Marshes, when combined with the predicted displacement from parts of Minsmere South Levels leads to a total loss of around 100ha of wetland foraging habitats (grazing marsh and reedbed), based on the WCS (although note our concerns around the visual disturbance buffer applied – see below).

3.400 Table 8.12 of the Shadow HRA Report Vol. 1 Part 1 estimates the percentage of foraging resource lost within various distance bands from the nest site at Minsmere. This concludes that around 20% of the total wetland habitats available within 4km of the nest site will be lost due to

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209 Use of the word "buffer" is perhaps misleading since it is used to describe the area in which impacts are assumed to occur so Ruddock and Whitfield (2007) are actually saying that *impacts would occur* within 300- 500 metres whereas the Applicant is saying impacts would only extend to 150 metres. At 150 metres this means that the area assumed to be affected by visual disturbance is within the area also assumed to be affected by noise disturbance.

210 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

211 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 4 of 5 (APP-148)

212 Ruddock, M. & Whitfield, D.P. (2007) A review of disturbance distances in selected bird species, Natural Research (Projects) for SNH

213 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

214 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

displacement during the construction phase, or 23% of total foraging resource available if also considering loss of agricultural habitats.

Lack of compensation for loss of arable habitats

3.401 Arable habitats near nest sites can be very important for foraging marsh harriers, especially male birds. This is illustrated by paragraph 6.3.94 and Table 8.12 in the Shadow HRA Report Vol. 1 Part 1[215] which discuss marsh harrier flight activity intensity. It is notable that some of the agricultural areas have levels of flight activity that are comparable to parts of the wetland habitat at Sizewell Marshes. Table 8.12 shows that 24% of the arable foraging area within 4km of the nesting area will be 'lost' to harriers during construction, yet the calculation of the foraging area required to mitigate lost foraging areas only considers wetland habitats. While we recognise that wetland habitats may represent optimal foraging areas and are more limited in availability, the loss of arable foraging is likely to be important to this population due to the level of use of these areas shown by the Applicant's surveys and based on the literature discussed in the section above. We therefore disagree with the statement in the Shadow HRA Report Volume 4: Compensatory Measures[216] paragraph 1.3.6 that arable habitats provide low quality foraging and its loss is unlikely to be limiting in this area.

3.402 We welcome the update to the marsh harrier baseline surveys carried out in 2020 and outlined in paragraphs 6.3.31 – 6.3.33 of the Shadow HRA Report Addendum[217], noting there are some slight changes to a few viewpoint locations. We note that the outputs of these surveys are presented using the flight intensity metric (m/hr/ha) and that paragraph 6.3.54 states again that the flight intensity over agricultural land is lower than that over Sizewell Marshes and the South Levels. However, it can be seen that when areas not used at all (possibly relating to crop type) are excluded, levels at other viewpoints are similar to South Levels and Marshes. This lends further support to our concern about the lack of consideration of arable habitats in the compensation provision.

3.403 We are therefore concerned with the statement in paragraph 8.8.259 of the Shadow HRA Report (and echoed in paragraph 3.4.13 of the Shadow HRA Report Vol. 4: Compensatory Measures), which states that "foraging birds would be able to counteract the potential 'loss' of the foraging resource on arable habitats… via behavioural mechanisms, such as modifying foraging ranges to exploit alternative areas of arable habitat … or by increasing foraging effort." We do not agree that marsh harriers will simply be able to increase foraging effort to counter the loss of arable habitats. This would involve an energetic and time cost which could result in reduced prey delivery rates and potentially affect the species' productivity.

3.404 We therefore disagree with the approach of excluding the loss of foraging resource from all agricultural habitats from the calculation of the compensation requirement.

Visual disturbance

3.405 As discussed above, we are concerned that the Applicant's use of a 150m buffer in the assessment of displacement arising from visual disturbance risks under-estimating impacts. Paragraph 8.8.206 of the Shadow HRA Report Vol. 1 Part 1[218] identifies 300 – 500m as an adequate buffer from human disturbance for nest sites and paragraphs 8.8.204 – 8.8.205 also discuss studies identifying an adverse impact on habitat use by marsh harriers from human disturbance. We

[215] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
[216] Shadow HRA Report Volume 4: Compensatory Measures (APP-152)
[217] Shadow HRA Report Addendum (AS-173)
[218] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

therefore consider that this potential underestimation of impacts may mean that the compensatory habitat requirement is also underestimated.

Compensatory requirement based on levels of flight activity

3.406 Table 8.14 in the Shadow HRA Report Vol. 1 Part 1[219] shows the results of an analysis of the foraging habitat resource lost to displacement based on estimated flight activity within the affected areas. This applies data from the baseline surveys of flight activity to all similar habitats within the same distance band from the nest site at Minsmere to estimate the percentage reduction in flight activity of foraging marsh harrier arising from displacement. It is estimated that 15% of the flight activity over wetland habitats within 4km of the nest site would be lost and that this would rise to 19-20% of total flight activity when considering both wetland and agricultural habitats.

3.407 We note our concern that compensatory habitat requirements should not be calculated solely on the basis of the flight activity. Whilst the level of flight activity may give some indication of the importance of an area for foraging, it does not distinguish between commuting and foraging flights and does not give an indication of the hunting success rate or level of biomass provided to foraging harriers. We therefore consider that greater weight should be given to habitat extent and quality in assessing the compensatory requirements.

Residual loss of foraging resource

3.408 Table 8.15 in the Shadow HRA Report Vol. 1 Part 1[220] indicates the different levels of wetland resource lost given different levels of increased marsh harrier activity over the compensation area and varying levels of displacement/barrier effect. We are concerned about the accompanying statement (paragraph 8.8.254) that "small" residual losses of total wetland foraging resource of less than 5% would be unlikely to have a population level impact, given the total wetland resource supports 50% of the SPA population. Given the difficulty likely to be encountered in assessing performance of the compensation area (as replacement of nutritional provision may not directly equate to flight activity), the precautionary principle must be applied.

3.409 We therefore do not agree with the statement in paragraph 3.4.10 of the Shadow HRA Report Vol. 4: Compensatory Measures[221]) that:

> *"If displacement and (particularly) barrier effects operate at less than 100% then the overall functional replacement (and therefore compensation) requirement would also be reduced."*

3.410 It would not be appropriate to take active steps to reduce the extent of the compensation area in this instance due to our concerns that the compensation area is not adequate in terms of extent or likely prey provisioning and the possibility that marsh harrier behavioural response may change again in future due to changing construction activities. In addition, again in line with the precautionary principle, should there be any doubt as to the effectiveness of the compensation area, more rather than less compensation should be provided. The EC guidance on "Managing Natura 2000 sites" (EC, 2018)[222] explains that the extent of compensation required is directly related to:

- the quantitative and qualitative aspects inherent to the elements of integrity likely to be impaired

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219 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
220 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
221 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)
222 European Commission (2018) Managing Natura 2000 sites. The provisions of Article 6 of the 'Habitats' Directive 92/43/EEC

- the estimated effectiveness of the measure(s)

3.411 EC (2018) also states that ratios are best set on a case-by case basis, but should generally be well above 1:1. Ratios of 1:1 or below should only be considered when it can be shown that measures will be fully effective in reinstating structure and functionality in a short period of time.

Ecological considerations for design of the compensatory marsh harrier foraging habitat

3.412 The Applicant has proposed the management of an area of 48.7ha of former arable land close to the Minsmere South Levels for foraging marsh harrier, with the aim of providing compensation for loss of foraging resource through a significant uplift in prey abundance and availability at this site. The following sections discuss the design of the proposed compensatory habitats and assess the adequacy of the compensation in comparison to the predicted impacts.

Habitat preferences of foraging marsh harriers

3.413 Much of the literature regarding marsh harrier foraging habitat preference recognises that whilst wet habitats are preferred by marsh harriers for nesting and are heavily used for foraging (particularly by females), arable farmland is also regularly used by foraging (and in some areas, nesting) harriers. The level of use of arable habitats appears to depend partly on its proximity to the nest site, with areas within 3km of the nest being most heavily used, and partly on the availability and accessibility of prey (Cardador and Mañosa, 2011[223]).

3.414 Prey accessibility is an important consideration in arable habitats (Schlaich *et al.*, 2015)[224], with some crops holding relatively good prey populations but having a level of vegetation cover during the breeding season which precludes hunting by marsh harriers (although these habitats may be suitable for foraging when the crop is at different vegetative stages at other times of year). For example, several studies report that maize fields are avoided by foraging harriers during the breeding season probably because the vegetation is too high and the cover too dense for harriers to hunt successfully, although they may be used outside the breeding season (Alves *et al.* 2014[225], Cardador and Mañosa 2011, Cardador *et al.* 2011[226]). Cardador *et al.* (2011) and Cardador and Mañosa (2011) note that a similar pattern of use was found for wheat and barley fields in their studies of the Ebro Delta in Spain, with use of these fields largely in the winter and spring before dense cover develops.

3.415 A number of studies report that alfalfa (lucerne) fields provide high quality foraging habitat for marsh harriers (and other birds of prey) throughout the breeding season (e.g. Cardador and Mañosa 2011, Cardador *et al.* 2011, Schlaich *et al.* 2015). This is thought to be partly due to prey abundance as alfalfa supports good numbers of small mammals and attracts insects which in turn attract small birds, and partly due to prey accessibility; alfalfa requires regular mowing which means that the crop does not get too tall and dense. Similarly, Clarke (1995)[227] reports an example from Norfolk of marsh harriers foraging over pea fields, hunting starlings attracted by the large number of insects present.

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223 Cardador, L. & Mañosa, S. (2011) Foraging Habitat Use and Selection of Western Marsh-Harriers (*Circus aeruginosus*) in Intensive Agricultural Landscapes, Journal of Raptor Research 45(2): 168-173

224 Schlaich, A.E., Klaassen, R.H., Bouten, W., Both, C. & Koks, B.J. (2015) Testing a novel agri-environment scheme based on the ecology of the target species, Montagu's Harrier *Circus pygargus*. Ibis 157: 713-721

225 Alves, M., Ferreira, J., Torres, I., Fonseca, C. & Matos, M. (2014) Habitat Use and Selection of the Marsh Harrier *Circus aeruginosus* in an Agricultural-Wetland Mosaic. Ardeola. 61(2): 351-366

226 Cardador, L., Carrete, M. & Mañosa, S. (2011) Can intensive agricultural landscapes favour some raptor species? The Marsh harrier in north-eastern Spain. Animal Conservation 14(4): 382-390

227 Clarke, R. (1995) The Marsh Harrier. Hamlyn Species Guides. London

3.416 Schlaich *et al.* (2015) report on agri-environment scheme options aimed at Montagu's harrier (but also used by marsh harrier) in the Netherlands, which comprise strips of alfalfa interspersed with strips of set-aside (cereals, grasses and herbs).

3.417 The success of these areas was thought to be due to the high numbers of small mammals supported by the set-aside, which were made accessible to harriers following the cutting of the alfalfa. This corresponds with studies of small mammal diversity in different arable habitats by Aschwanden *et al.* 2007[228] and Scott *et al.* 2008[229], from which it can generally be stated that small mammals are most abundant in tall (unmown) grass and mosaic habitats. Clarke (1995) also notes the benefits of linear features such as ditches and edges between taller and shorter vegetation as these aid the element of "surprise" used by marsh harriers in hunting.

3.418 Male and female marsh harriers show different foraging habitat preferences. Males typically take smaller prey than females and forage more widely (Clarke, 1995 and Witkowski, 1989[230]). They also make more use of arable areas than females, with females being more likely to forage on wetland (Clarke, 1995, and Witkowski, 1989), including reedbeds. Females are also less agile than males so prefer the taller vegetation of wetlands where they can use the element of surprise in their hunting (Clarke, 1995). This may mean males are more likely to use the dry habitats within the proposed compensation area than females. Therefore, later in the breeding season when females start to forage, these birds could be more affected by the reduction in wetland foraging habitat available as they may be less willing to use grassland foraging areas.

3.419 We query therefore whether there could there be a resulting reduction in food supply for older (unfledged) chicks. This issue is acknowledged in Paragraph 6.3.88 of the Shadow HRA Report Vol. 1 Part 1[231], which briefly discusses the potential for male and female marsh harriers to exploit prey resources differently during the breeding season and the need to consider this in the provision of foraging habitat compensation.

Habitat components of the proposed compensation area

3.420 Sections 2. 1 And 2.2 and Table 1.1: Habitat components of the preferred compensation option(s) of the Shadow HRA Report Vol. 4: Compensatory Measures[232] summarise the principles of design and habitat components of the proposed marsh harrier compensation area. We agree with the principles set out in 2.1.2 and our comments on the various habitat components proposed are provided below.

Short grassland

3.421 Whilst we acknowledge that short grassland can be productive in terms of numbers of rabbits, we are concerned that marsh harriers may not make significant use of such areas. At Minsmere it is notable that marsh harriers rarely forage over the acid grassland, despite these areas holding the highest densities of rabbits, perhaps due to the difficulty of 'surprising' prey in these areas where the sward height is maintained below 10cm to provide suitable habitat for stone-curlew and woodlark. Areas of short grassland should be limited in extent, and the expansion of rabbits

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228 Aschwanden, J., Holzgang, O. & Jenni, L. (2007) Importance of ecological compensation areas for small mammals in intensively farmed areas. Wildlife Biology 13(2): 150-158

229 Scott, D., Joyce, C. & Burnside, N. (2008) The influence of habitat and landscape on small mammals in Estonian coastal wetlands. Estonian Journal of Ecology 57: 279-295

230 Witkowski, J. (1989) Breeding biology and ecology of the Marsh Harrier *Circus aeruginosus* in the Barycz valley, Poland. Acta Ornithologica 25(3): 223-320

231 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145), page 157

232 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

monitored (and if necessary, managed) to ensure that areas intended to support taller vegetation are not overgrazed.

3.422 It is also worth noting that rabbit numbers have crashed in many parts of the country, probably as a result of disease (in particular, a new strain of viral haemorrhagic disease) and so are not necessarily a reliable food source. This does not appear to have happened in the Minsmere area so far but could do, and in this case, it would be unlikely that habitat improvements could result in a significant uplift to population numbers.

Tussocky grassland

3.423 We agree that, with regard to dry habitats, tussocky grassland is a useful element to include due to the relatively large numbers of small mammals it is likely to support. A review of small mammal populations in the UK by Mathews *et al.* (2018)[233] found that unimproved grassland (here equivalent to rough or tussocky grassland) supported high densities of rabbits and field voles in particular. Studies by Aschwanden *et al.* 2007 and Scott *et al.* 2008[234] also found that, in farmland areas, small mammals are most abundant in tall (unmown) grass and mosaic habitats.

Scrub foci

3.424 We acknowledge the potential benefits of scrub foci to aid establishment of rabbits in short grassland and support its provision in small patches, as recommended in the report. We would not support the inclusion of larger patches of scrub/woodland as prey species using them may not be available to harriers.

Hedges and banks

3.425 We agree that providing some 'edge habitats' between vegetation of different heights in the form of hedges and earth banks could be useful as these may aid the element of "surprise" used by marsh harriers in hunting (Clarke, 1995)[235]. Edges between mown and unmown strips of grassland could also be beneficial.

Wild bird seed and nectar mixes

3.426 As discussed above, Schlaich *et al.* (2015)[236] found that strips of set-aside (cereals, grasses and herbs) provided good numbers of small mammals which were made accessible to harriers following the mowing of adjacent strips. We therefore support the inclusion of these habitats, although request further detail as to how these would be managed to maximise prey availability.

Wet woodland and reedbed

3.427 Paragraph 2.2.1-3 of the Shadow HRA Report Vol. 4: Compensatory Measures[237] explains that approximately 0.7ha of wet woodland and 1.2ha of reedbed habitat will be included within the marsh harrier compensation area. Whilst we welcome in principle the inclusion of reedbed habitat in the proposals for habitat compensation, we are concerned that this area is not due to be constructed until the first winter of the construction period. This means it is unlikely to be

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233 Mathews, F., Kubasiewicz, L.M., Gurnell, J., Harrower, C.A., McDonald, R.A. & Shore, R.F. (2018) A Review of the Population and Conservation Status of British Mammals. A report by the Mammal Society under contract to Natural England, Natural Resources Wales and Scottish Natural Heritage. Natural England, Peterborough

234 Scott, D., Joyce, C. & Burnside, N. (2008) The influence of habitat and landscape on small mammals in Estonian coastal wetlands. Estonian Journal of Ecology 57: 279-295

235 Clarke, R. (1995) The Marsh Harrier. Hamlyn Species Guides. London

236 Schlaich, A.E., Klaassen, R.H., Bouten, W., Both, C. & Koks, B.J. (2015) Testing a novel agri-environment scheme based on the ecology of the target species, Montagu's Harrier *Circus pygargus*. Ibis 157: 713-721

237 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

functional during the early phases of construction during which noise levels are highest and therefore its construction at this stage could effectively result in a loss to the compensatory habitat provided. This component therefore should be constructed prior to the main construction period commencing or be removed from the calculation of the area of compensatory habitats for marsh harrier.

3.428 The Marsh Harrier Mitigation Area Feasibility Report (Wood, 2018) (ES Vol 2 Ch. 14 Appendix 14C5[238]) acknowledges that trees and woodland should be avoided as prey they may provide would not be accessible to harriers. We therefore do not agree that the 0.7ha of wet woodland proposed can be counted towards marsh harrier compensation.

Open water (water resource storage area)

3.429 Paragraph 2.1.5 of the Shadow HRA Report Vol. 4: Compensatory Measures[239] explains the "inclusion of a water resource storage area in the north-eastern area of the 48.7ha [compensation] site". Based on the description in paragraphs 3.4.133-138 of ES Vol. 2 Ch. 3 Description of Construction[240], this will comprise a reservoir with raised embankments of up to 3m and surrounded by tree/hedge planting. This type of structure is not likely to provide suitable foraging habitat for marsh harriers unless significant amounts of emergent vegetation develops attracting wildfowl use. This is considered unlikely to occur unless relatively large areas of shallow sloping edge are provided and will not establish immediately even if the design is suitable. We therefore do not agree that this area can be included in the area proposed as compensation for marsh harrier foraging.

Conclusions regarding habitat components of the compensation area

3.430 In general, we agree with the dry habitat components proposed, although we still have concerns around the overall loss of valuable wetland foraging habitats for marsh harrier. The wetland habitats additionally proposed at present are unlikely to address this, either because the habitat types are of limited usefulness to foraging harriers (e.g. wet woodland) or are unlikely to be established before construction starts (e.g. reedbed), however, the inclusion of ditches would be likely to be beneficial.

3.431 Based on the information discussed above, in order to maximise abundance and accessibility of small mammals and birds in the dry habitats as prey for marsh harriers, strips of taller vegetation providing optimal habitat for small mammals along with strips of mown/shorter vegetation allowing access to foraging harriers should be considered, along with flowering crops which may attract small birds foraging for insects. These habitats should form a mosaic to produce "edges" between taller and shorter vegetation and should include other linear features, such as hedges and ditches. Consideration must be given to the management of the area to maintain a balance between the needs of rabbits and other small mammals and birds e.g. to avoid detrimental effects of overgrazing by rabbits during the establishment of seed mixes. Careful consideration must also be given to other potential conflicts e.g. effects of any mowing/cutting on ground nesting birds. The use of insecticides in the compensation area should be avoided.

3.432 Further details must be provided in a comprehensive management plan for the compensation site including details of how management will be carried out and performance monitored.

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238 ES Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology Appendix 14C5 Marsh Harrier Mitigation Area Feasibility Report (APP-259)

239 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

240 ES Vol. 2 Main Development Site Ch. 3 Description of Construction (APP-184)

Marsh harrier prey species

3.433 Paragraphs 3.4.1 to 3.4.4 of the Shadow HRA Report Vol. 4: Compensatory Measures[241] discuss the main components of marsh harrier diet. Table 3 below summarises a number of studies of marsh harrier diet in the breeding season. This supports the conclusions in the report that marsh harriers are able to take a range of prey species as opportunity allows.

Table 3 Prey types taken by marsh harriers

	Tornberg & Haapala, 2013[242] (Finland)	Cardador *et al.*, 2012[243] (Iberian peninsula)	Brzezinski & Zmihorski, 2009 (Poland)[244]	Witkowski, 1989[245] (Poland)	Underhill-Day, 1985[246] (East Anglia, UK)
Gamebirds	x			x	x
Domestic fowl			x		
Waders	x			x	x
Waterfowl	x		x	x	x
Gulls & terns	x			x	x
Small passerines			x	x	x
Unspecified birds		x			
Eggs				x	x
Hares	x		x	x	x
Rabbits					x
Moles			x	x	
Voles		x		x	x
Mice		x	x	x	
Rats		x		x	x
Other rodents	x			x	
Shrews		x		x	
Mustelids			x		x
Hedgehogs				x	
Carrion (mammal)				x	
Fish		x	x	x	
Amphibians			x	x	
Reptiles		x		x	
Invertebrates					x

241 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

242 Tornberg, R. & Haapala, S. (2013) The diet of the Marsh Harrier *Circus aeruginosus* breeding on the isle of Hailuoto compared to other raptors in northern Finland. Ornis Fennica 90(2): 103-116

243 Cardador, L., Planas, E., Varea A. & Mañosa, S. (2012) Feeding behaviour and diet composition of marsh harriers *Circus aeruginosus* in agricultural landscapes. Bird Study, 59(2): 228-235

244 Brzeziński, M. & Żmihorski, M. (2009) Nestling Diet and Parental Provisioning Behaviour in the Marsh Harrier (*Circus aeruginosus*). Acta Zoologica Lituanica. 19(2): 93-98

245 Witkowski, J. (1989) Breeding biology and ecology of the Marsh Harrier *Circus aeruginosus* in the Barycz valley, Poland. Acta Ornithologica 25(3): 223-320

246 Underhill-Day, J.C. (1985) The food of breeding of Marsh Harriers *Circus aeruginosus* in East Anglia. Bird Study 32(3): 199–206

3.434 While we therefore agree that the prey items likely to be provided by the compensatory habitats would be suitable for marsh harriers, we reiterate our concern that the overall loss of wetland foraging habitat could negatively affect prey provisioning later in the breeding season when females also bring prey to the chicks. As discussed in section above and noted in several of the studies summarised in table 3, females typically take larger prey than males and forage proportionately more in wetland habitats than males. Therefore, the loss of wetland habitat could increase intra-specific competition and reduce the biomass provided by females or increase female energetic costs in order to provide similar levels of biomass.

Predictions of prey abundance

Rabbits

3.435 Section 3.2 of the Shadow HRA Report Vol. 4: Compensatory Measures[247] discusses predictions of the prey abundance which could be provided by the compensatory habitats and draws on the information provided in Appendices B of ES Vol. 2 Ch. 14 Appendix 14C5 Marsh Harrier Mitigation Area Feasibility Report[248] ("Appendix B"). Appendix B concludes that the compensation area is likely to have supported rabbits at a density of 0.5-1 per hectare during agricultural use. This is based on average figures of 2.5 per hectare for arable areas from Harris (1995)[249] and the statement that "Arcadis however has reported virtually never seeing a rabbit within the proposed mitigation area during site visits".

3.436 Given that rabbits are primarily active at night (we assume Arcadis' visits were carried out during the day) and that surveys specifically looking for rabbits at night may only detect around 60% of the rabbit population (Matthews *et al.*, 2018), this is not sufficient justification to reduce the estimated density of rabbits from the recognised UK sources of such information (Harris, 1995, also cited in the review by Matthews *et al.*, 2018). Whilst it is acknowledged that limited evidence is available for rabbit densities in the UK and that densities are known to be highly variable, it should also be considered that the area is on sandy soils which are known to support rabbit densities at the higher end of such estimates (Cowan, 1991, cited in Harris, 1995).

3.437 Appendix B goes on to discuss the densities of rabbits that may be possible on the compensation area once the new habitats are established. Based on statements that Harris (1995) reports a range of rabbit abundances and that there are currently high (but unquantified) densities of rabbits in neighbouring areas, it is assumed that winter densities of 15-20 rabbits per hectare would be achievable (higher in the breeding season) on the short grassland and 2.5-5 per hectare on the tussocky grassland.

3.438 This does not accord with the estimated densities of 2.5 per hectare on lowland grassland used by the Harris (1995) review or the Statutory Nature Conservation Bodies' more recent review (Matthews *et al.*, 2018) which quote densities of between around 0.5 and 6 per hectare for improved grassland and 5 per hectare for unimproved grassland. It is stated that the abundance of rabbits will increase by 15-40 times in the short grassland compared to baseline and by 2.5-10 times on the tussocky grassland.

3.439 Therefore, the Applicant's estimates appear extremely optimistic when compared to the figures in the two reviews cited and we do not agree that any confidence can be attached to these figures. Whilst we acknowledge the limitations of the evidence available from the literature, we do not

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247 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

248 ES Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology Appendix 14C5 Marsh Harrier Mitigation Area Feasibility Report (APP-259)

249 Harris, S., Morris, P., Wray, S. & Yalden, D. (1995) A review of British mammals: population estimates and conservation status of British mammals other than cetaceans. JNCC, Peterborough

agree that any site-specific evidence has been presented which would justify such a significant departure from the literature. We therefore consider that the figures in Matthews *et al.* (2018) which are based on Harris (1995) and others should be used and would indicate that the original arable is likely to have supported around 2.5. rabbits per hectare and that the upper range figure of 6 per hectare for improved grassland could be achieved, equating to an increase of 2.4 times the original density.

Total small mammal numbers

3.440 Appendix B also calculates the numbers of small mammals provided by the baseline habitats and the various habitat options for the compensation area. It provides summaries of a number of studies relating to small mammal numbers in various habitats, but most provide information regarding the relative importance of habitats for small mammals, with only one summary including figures for mammal densities which could be used in the metric to calculate densities for the compensation area.

3.441 The key review of small mammal population in the UK by Mathews *et al.* (2018)[250] for the Statutory Nature Conservation Bodies has not been included, and as the most up to date and authoritative recent review relevant to the UK we query why this is the case. Mathews *et al.* (2018) provides density estimates for various habitats for the key small mammal species – this is discussed further below. A brief summary of a study of small mammals in the compensation area is also provided. Whilst this provides useful information (e.g. regarding the species present) and some estimates of density, it is noted that the data are limited. No information is provided regarding these data limitations which would allow the robustness of the data to be assessed and we recommend that a report of this study is provided.

3.442 The report then goes on to suggest mammal densities for habitats within the compensation area for use in calculating a metric to compare the numbers of small mammals likely to be found in the compensatory habitats to those in the baseline habitats. Very little justification is provided for the figures used.

3.443 We have compared these figures with the density figures provided in the UK review by Mathews *et al.* (2018) in Table 4 below based on all the small mammal species expected to be present in the area for which data are available. We would also expect that weasel, brown rat and harvest mouse could also be present, however, Mathews *et al.* (2018) conclude that there is too little evidence available to support density figures for these species and habitats.

[250] Mathews, F., Kubasiewicz, L.M., Gurnell, J., Harrower, C.A., McDonald, R.A. & Shore, R.F. (2018) A Review of the Population and Conservation Status of British Mammals. A report by the Mammal Society under contract to Natural England, Natural Resources Wales and Scottish Natural Heritage. Natural England, Peterborough

Table 4 Densities of small mammals for habitats in the marsh harrier foraging area – comparison of densities derived from Mathews *et al.* (2018) with those in the Marsh Harrier Mitigation Area Feasibility Report Appendix B (figures presented are median densities, with 95% confidence intervals in brackets, where available)

Species/group	Arable (ha^{-1})	Short (improved) grassland (ha^{-1})	Tussocky (unimproved) grassland (ha^{-1})	Hedgerows (km^{-1})	Wildbird seed and nectar flower mixes (ha^{-1})
Figures from Mathews *et al.* (2018)					
Hedgehog	0.05	<0.01	0.4	n/a	*0.4*[+]
Mole	1.30	2.65	4.0	n/a	*4.0*[+]
Common shrew	n/a	0.10 (0 - 0.50)	4.80 (4.0 – 9.50)	7.70 (2.07 – 13.9)	*4.80 (4.0 – 9.50)*[+]
Pygmy shrew	n/a	0 (0 - 0.1)	4.80 (1.20 – 4.80)	0.1 (0.01 – 30)	*4.80 (1.20 – 4.80)*[+]
Water shrew	n/a	n/a	0.5	n/a	*0.5*[+]
Rabbit	2.5	0.48 (0.26 – 0.6)	5	n/a	*5*[+]
Brown hare	0.11 (0.08 – 0.3)	0.04 (0.03 – 0.26)	n/a	n/a	*n/a*[+]
Bank vole	n/a	0.1 (0 – 1)	1 (0 – 3)	10.5 (0 – 26)*	*1 (0 – 3)*[+]
Field vole	n/a	n/a	48 (33.33 – 52.5)	7.5 (2 – 52)*	*48 (33.33 – 52.5)*[+]
Wood mouse	n/a	0.1 (0 – 0.5)	6.5 (0 – 13)	3 (0 – 12)*	*6.5 (0 – 13)*[+]
Yellow-necked mouse	n/a			8.5 (6 – 11)	
Stoat	0.02	0.01	0.06	n/a	*0.06*[+]
TOTAL	3.98	3.48	75.06	37.3	*75.06*[+]
Figures from Marsh Harrier Mitigation Area Feasibility Report Appendix B					
All small mammals (including rabbits)	0 - 21	10 – 35	102.5 - 305	500 – 1000 ha^{-1}	50 - 101

[+]No direct figures for this habitat are available from Mathews *et al.* (2018), but as habitat structure is key and this is most similar to longer (Mathews *et al.*'s unimproved) grassland, these figures are tentatively applied

* Figures are for hedgerows in agri-environment schemes

3.444 From the table, it can be seen that the numbers used in Appendix B for some habitats, particularly tussocky grassland and hedgerows, appear high in comparison to Mathews *et al.* (2018), particularly as the assessments go on to use the highest figures from the ranges presented (e.g. 300 per hectare for tussocky grassland).

3.445 It should be noted that Mathews *et al.* (2018) explain that in many instances the data regarding density are limited and hence their estimates for different species and habitats may vary in their reliability. Also, the lack of data for some species, particularly brown rat, will mean that the total figures above are underestimates, but it appears unlikely that this would account for the significant discrepancy with the figures used in Appendix B.

3.446 Table 5 below compares the habitat areas for the compensation area under baseline conditions and the preferred compensation option 2b (note scrub foci are excluded as the areas are so small). These are then used with the density figures from Table 4 above to calculate a comparison

(in Table 6) of small mammal numbers within the compensation area using the figures in Appendices B (noting that outputs are presented in Appendix C) and those of Mathews *et al.* (2018).

Table 5 Comparison of habitat areas under baseline conditions and for compensation option 2b

	Baseline (ha)	**Compensation option 2b (ha)**
Arable	39.4	0
Short grassland	1	8
Tussocky grassland	0.71	24
Hedgerow	0.5 (1650m)	6.95 (6320m)
Wildbird seed and nectar mix	5.35	8

Table 6 Comparison of projected small mammal numbers (including rabbits) based on Appendices B and C of the Marsh Harrier Mitigation Area Feasibility Report and Mathews *et al.* (2018) for baseline and compensation option 2b habitats

	Baseline – using Appendix B max density figures	**Baseline – based on Mathews *et al.* (2018)**	**Compensation option 2b – using Appendix B max figures**	**Compensation option 2b – based on Mathews *et al.* (2018)**
Arable	827	156.8	0	0
Short grassland	35 (App C: 40)*	3.48	280 (App B: 320)*	27.8
Tussocky grassland	217	53.3	7320	1801.4
Hedgerows	500 (App C: 495)	61.5	6952	156.3
Wildbird seed and nectar mixes	540	401.6	808	600.5
TOTAL small mammals	2119	676.7	15360 (App B: 15400)	2586
Uplift compared to baseline using same source of density figures			x7.25	x3.82

*Note there are some discrepancies in the density figures used for rabbits in Appendix B – although the density stated for short grassland is 10-15 per hectare, the calculations use 15-20 per hectare

3.447 From the above it can be seen that the high small mammal densities applied have the effect of significantly overestimating small mammal abundance for the compensatory habitats and exaggerating the uplift from baseline conditions.

3.448 We therefore disagree with the statements in Section 3.2 of the Shadow HRA Report Vol. 4: Compensatory Measures[251] regarding the level of uplift in small mammal abundance that can be expected from the habitat compensation area. Paragraph 3.2.3 states that this is expected to be between 6.6 and 10.7 relative to baseline conditions, but whilst noting there are limitations to

[251] Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

the figures discussed above, we consider that the uplift in abundance is likely to be around half of this level.

3.449 We also note that whilst many of the potential prey species are primarily herbivorous, not all of them are, and their food supply also needs to be considered. Given the arable history of this land and likely associated chemical use, we can expect invertebrate populations to be significantly depressed and unlikely to support an immediate uplift in insectivorous species. The wider context of the ongoing decline in bio-abundance across all taxonomic groups across the UK described in the most recent State of Nature Report (Hayhow *et al.*, 2019[252]), may also make this unrealistic.

Effect of water resource storage area and wetland habitats

3.450 Paragraph 3.2.2 of the Shadow HRA Report Vol. 4: Compensatory Measures[253] states that the additional benefits of the water storage area and wetland habitats have not been considered and that these are expected to enhance they prey resource. As discussed above, this would rely on these habitats being functional by the beginning of the construction period, otherwise they represent a loss from the compensatory habitat.

Predictions of uplift in flight activity

3.451 At present, 48.7ha of habitat is proposed to replace a potential loss of around 100ha of foraging habitat, on the assumptions that it is closer to the nesting area than Sizewell Marshes, and that it is possible to increase resource provision sufficiently to attract greater levels of activity. Paragraph 8.8.249 of the Shadow HRA Report Vol. 1 Part 1[254] explains that when taking into account loss of wetland habitat only, the level of flight activity required at the compensation site will be 18-19 m/ha/hr which is greater than average levels on Sizewell Marshes and parts of Minsmere South Levels.

3.452 We are also concerned that loss of arable foraging habitat has not been included in the calculation of the compensation requirements. Paragraph 8.8.257 explains that if loss of arable foraging were to be considered alongside loss of wetland, the resulting level of flight activity required at the compensation site to account for this would be 34-35 m/ha/hr. It is acknowledged that this would be close to or above the highest level of activity on the baseline surveys and was observed over a small area of the Minsmere South Levels only.

3.453 We therefore do not agree that the level of uplift in flight activity likely to be achievable will be comparable with that required based on this calculation (although note our concerns with reliance on this metric, discussed above).

Additional constraints on the functionality of the compensatory habitats

Additional visual disturbance constraints

3.454 We note that the footpath running from Eastbridge to Minsmere Sluice passes through the northern extremity of the proposed marsh harrier compensation area and therefore that visual disturbance from human activity along this route could compromise the function of the parts of the compensation area.

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252 Hayhow, D.B., Eaton, M.A., Stanbury, A.J., Burns, F., Kirby, W.B., Bailey, N., Beckmann, B., Bedford, J., Boersch-Supan, P.H., Coomber, F., Dennis, E.B., Dolman, S.J., Dunn, E., Hall, J., Harrower, C., Hatfield, J.H., Hawley, J., Haysom, K., Hughes, J., Johns, D.G., Mathews, F., McQuatters-Gollop, A., Noble, D.G., Outhwaite, C.L., Pearce-Higgins, J.W., Pescott, O.L., Powney, G.D. & Symes, N. (2019) The State of Nature 2019. The State of Nature partnership

253 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

254 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

Noise levels in compensation area

3.455 As acknowledged in paragraphs 8.8.188 and 8.8.195-8.8.197 of the Shadow HRA Report Vol. 1 Part 1[255], noise levels in parts of the marsh harrier compensation area itself will also exceed the 70dB L_{Amax} threshold for all construction phases, except Phases 3 and 4, and we are therefore concerned that this could limit the functionality of the compensation area through avoidance by marsh harrier and/or avian prey species.

3.456 It is also not clear how the overlap of construction phases in some years will affect this assessment. It is explained that higher levels of noise within the compensation area (over 70dB L_{Amax}) are likely to occur for limited time periods and over limited spatial extents, however, this does represent a further reduction in the functionality of the area, which even at its full extent, is significantly smaller than the area subject to displacement and barrier effects.

Additional stockpile

3.457 We note that paragraph 2.2.108 of Vol. 1 Environmental Statement Addendum Ch. 2 Main Development Site[256] refers to an additional stockpile up to 15m above ground level next to the proposed marsh harrier compensation area and query whether the potential for additional noise impacts on the compensation area has been considered.

Water resource storage area and flood mitigation measures

3.458 Paragraph 8.3.22 of the Shadow HRA Report Addendum[257] states that

> *"The differences in the Phase 1 modelling associated with this area are not relevant to breeding marsh harrier because the construction of the water storage area would occur during the first winter of the construction period. As such, the noise disturbance from these works would not coincide with the seasonal period relevant to the SPA breeding marsh harrier population."*

3.459 Based also on Figure 8A.1 of the Shadow HRA Report Addendum Appendices 1A to 10A Part 5[258], construction of the water storage area in the first winter of construction phase 1 would cause the 70dB L_{Amax} contour to affect the southern part of the North Levels (part of the core part of RSPB Minsmere) – this is a significant northward extension of this contour as a result of the proposed change. If this work were to stretch into the breeding period, the 65dB L_{Amax} contour stretches even more significantly into the North Levels reedbeds and has potential for significant impacts on breeding marsh harrier and breeding bittern.

3.460 It should be noted that Clarke (1995)[259] states that marsh harriers take up their breeding territories in March and that courtship commences immediately while bitterns typically start booming in February and noise disturbance could have the potential to interfere with this.

3.461 In addition, the vast majority of the marsh harrier compensation area is also within the 70dB L_{Amax} contour meaning that should such impacts extend into the breeding season, the compensation would not function. Given the timing of these works is therefore crucial to the conclusion of no additional impact, we query how will these timing restrictions be confirmed and secured.

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255 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
256 Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)
257 Shadow HRA Report Addendum (AS-173)
258 Shadow HRA Report Addendum Appendices 1A-10A Part 5 (AS-178)
259 Clarke, R. (1995) The Marsh Harrier. Hamlyn Species Guides. London

3.462 We also query the expected noise contours during the breeding season in Phase 1 using the updated modelling. The updated figures include the first winter of Phase 1 and separate figures for Phase 2 – they do not account for the breeding season in Phase 1 or the fact that the construction phases are expected to overlap (see Plate 3.1 in ES Vol. 2 Ch. 3 Description of Construction[260]). Combined WCS maps where construction phases are expected to overlap and impacts would be worse than each construction phase treated alone should be provided to avoid under-estimation of impacts.

3.463 Paragraphs 2.2.121 and 2.9.53 of Vol. 1 Environmental Statement Addendum Ch. 2 Main Development Site[261] explain that the water storage area's original proposed location will now be used to create open water and wet reedbeds which will provide foraging habitat for marsh harriers. But, in addition to the concerns above regarding any slippage in the timing of the excavation works (noting this is proposed for the early stages of construction Phase 1), we also note that these habitats may take some time to establish and hence may not be functional (in terms of prey provision for marsh harriers) during key phases of construction.

3.464 This area could therefore be viewed as a loss to the area of compensation provide until these habitats become functional, therefore we recommend that any wetland habitat is created in advance of the main construction period.

Timing of functionality

3.465 With regard to the marsh harrier compensation site, paragraph 2.4.1 of the Shadow HRA Report Vol. 4: Compensatory Measures[262] states that:

> *"The proposed habitat enhancement land was taken out of agricultural production approximately 4 years ago and some habitat management – for the purposes of offsetting the effects of the Sizewell C Project – has been implemented in the intervening period and is ongoing. Further habitat enhancement, including scrub and hedgerow planting was undertaken in early 2020."*

3.466 No further details of the works already carried out are provided and it is not clarified as to whether the works carried out so far are in line with the management recommendations in ES Vol. 2 Ch. 14 Appendix 14C5 Marsh Harrier Mitigation Area Feasibility Report[263].

3.467 Paragraph 2.4.4 states that the habitats will be functional by the time construction commences due to being out of agricultural production for 7 years, however, it will be necessary to demonstrate that the habitat measures in place will be aligned with the agreed and finalised compensation plan and sufficiently established to provide the required levels of prey provisioning when construction commences. Further details of the habitat management carried out to date and any subsequent uplift in marsh harrier activity should therefore be provided.

3.468 We also query the construction timeline in Plate 2.1 of the Shadow HRA Report Vol. 1 Part 1[264], which shows that the marsh harrier habitats will be established during construction Phases 1 to 3. This appears to contradict the statements discussed above, but if Plate 2.1 is correct, this would be of significant concern as we understand that the early phases of construction will generate some of the highest noise levels and would therefore lead to unacceptable residual impacts. This

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260 ES Vol. 2 Main Development Site Ch. 3 Description of Construction (APP-184)
261 Volume 1 Environmental Statement Addendum Chapter 2 Main Development Site (AS-181)
262 Shadow HRA Report Volume 4: Compensatory Measures (APP-152)
263 ES Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology Appendix 14C5 Marsh Harrier Mitigation Area Feasibility Report (APP-259)
264 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

approach would also be contrary to the criteria for compensatory habitats set out in Section A.7 of Annex A of NPS EN-6 (Nuclear Power)[265], that they must be:

> *"capable of ensuring that the Natura 2000 site is not irreversibly affected by the project before the compensation is in place".*

3.469 Paragraph 8.8.244 of the Shadow HRA Report Vol. 1 Part 1 implies that the marsh harrier compensatory habitats will still be available for use by harriers during the decommissioning process. Whilst we would welcome this, we request that further detail is provided to clarify the intended approach and how the area will be managed during the construction period.

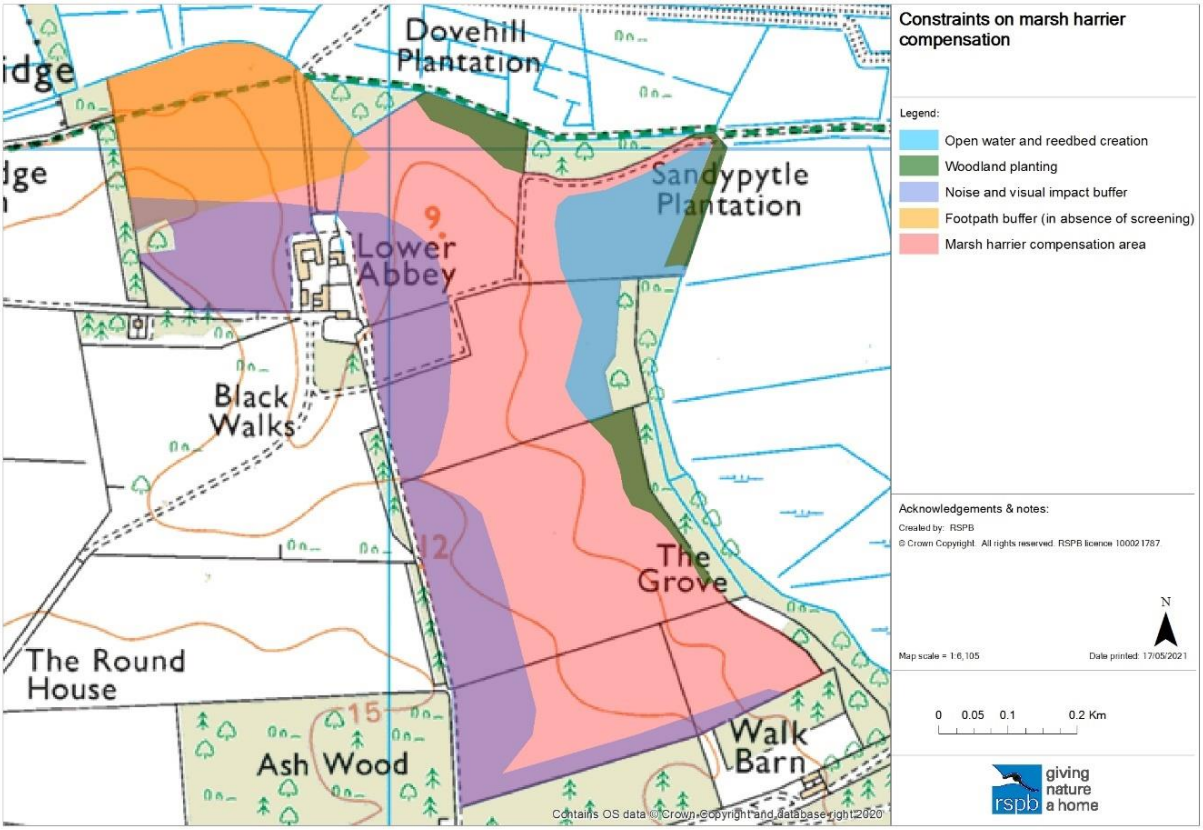
Summary of constraints on the functionality of the marsh harrier compensation area

3.470 As discussed above, there could be a number of constraints affecting the functionality of the marsh harrier compensation area, particularly during the first few years of construction. The map (Figure 1) shows the approximate area of the marsh harrier compensation area affected by the following:

- The maximum extent of the 70dB L_{Amax} noise contour and 150m visual disturbance buffer during the breeding seasons in the Phase 1 and 2 construction periods – based on Figure 8.10 of the Shadow HRA Report Vol. 1 Part 4[266]. Note that as discussed above, the maps based on the revised modelling in the Change Application are not clear for the first breeding season, or for the overlap between construction phases 1 and 2.
- The woodland planting illustrated in Figure 2.2.14 of Vol. 2 ES Addendum Ch. 2 Main Development Site Figures Part 1[267]
- The reedbed and open water creation also shown in Figure 2.2.14 as above. We would not expect this to be functional in terms of prey provision for marsh harrier during this period.
- A 150m buffer (based on the Applicant's chosen visual buffer, although note that this lacks precaution) around the footpath on the northern edge of the site, where this does not appear to be screened by woodland.

[265] National Policy Statement for Nuclear Power Generation (EN-6) Volume II Annex A
[266] Shadow HRA Report Vol. 1 Part 4 (APP-148)
[267] Vol. 2 ES Addendum Ch. 2 Main Development Site Figures Part 1 of 4 (AS-190)

Figure 1 Constraints on the functionality of the marsh harrier compensation area during the breeding seasons in construction phases 1 and 2.



3.471 It should be noted that this assumes that the planned winter construction of the wet habitats does not over-run into the following breeding season. If this were to occur, very little of the compensation area would remain outside the 70dB L_{Amax} noise contour.

3.472 Also we note that the planned woodland planting does have the effect of reducing the areas which would otherwise be affected by disturbance from the footpath on the northern edge of the site, however, as woodland is unsuitable for foraging marsh harriers, we still consider it should not be included in the calculations of compensatory habitat provision.

3.473 As a result of these constraints, we estimate that only around half of the marsh harrier compensation area will be functional during the early stages of construction. Given that we do not agree that the full area is adequate to fully compensate for the foraging resource lost, this significantly increases our concerns about unacceptable residual impacts based on the current proposals.

Conclusions regarding compensatory habitats proposed for loss of marsh harrier foraging resource

3.474 The points below relate to Table 1.3 of the of the Shadow HRA Report Volume 4: Compensatory Measures[268]) which compares the compensation measures proposed to the requirements of EN-6[269]. We address each of these requirements in turn.

[268] Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

[269] National Policy Statement for Nuclear Power Generation (EN-6) Volume II Annex A

Be appropriate for the area and the loss caused by the Sizewell C Project

3.475 We do not at present have confidence that the compensation area can sustain the necessary level of prey provision or harrier activity to fully address the loss of foraging resource caused by noise-induced displacement/barrier effects.

3.476 This is due to our concerns about noise levels in the current compensation area, its limited extent compared to the area to be lost (especially when also considering arable habitats) and doubts about the ability to achieve the level of uplift in prey provision and flight activity predicted.

3.477 We also disagree that the assessment of noise impacts involves a "high degree of precaution" as we note there are several areas of uncertainty (for example, the robustness of the 70dB L_{Amax} impact threshold) due to which it is appropriate that the assessment is precautionary.

Be capable of protecting the overall coherence of the Natura 2000 network

3.478 Based on the above, we do not agree at this stage that the compensation measures are sufficient to protect the overall coherence of the National Sites Network (formerly known as the Natura 2000 Network of European sites within the UK) for breeding marsh harriers.

Be capable of implementation

3.479 Whilst we agree it is likely that the dry habitats could be implemented, a detailed habitat management plan and a robust monitoring will be required in order to assess whether the habitats will be fully established and functional (in terms of prey provisioning) before construction starts and whether the proposed habitat management of the compensation area will maintain it in suitable condition for the required period.

Be capable of ensuring that the Natura 2000 site is not irreversibly affected by the Sizewell C Project before the compensation is in place

3.480 As above, it must be demonstrated that the habitats provided meet the specification set out in a detailed habitat management plan.

3.481 As discussed above we also have concerns about the likelihood of being able to establish the wet habitats before construction starts.

3.482 We also recommend that prey population levels are monitored to determine whether the habitats are performing as required in terms of prey availability and abundance (as discussed further below).

Be directed in measurable proportions to the habitats and species negatively affected

3.483 Whilst we agree that the proximity of the compensation area to Minsmere is beneficial, we do not agree that 48.7ha of dry habitats are adequate to compensate for the foraging provided by 100ha of wet habitats due to the differences in habitat extent and type and the likely effects on prey provisioning levels. We also are concerned that loss of access to arable habitats is not compensated.

Be related to the same biogeographical region (within the UK)

3.484 We agree and note the benefits of the proximity of the compensation site to the habitats effectively lost.

Serve functions that are comparable to those that motivated the original area's submission for designation

3.485 We agree that the compensatory habitats will provide a similar supporting function to the SPA to those habitats effectively lost through providing foraging opportunities for marsh harriers, however, we consider that this supporting function will be reduced compared to that provided by the area lost.

Be clearly defined, with implementation goals and managed so that the compensatory measures can achieve the goal of maintaining or improving the overall coherence of the National Sites Network (formerly known as the Natura 2000 network)

3.486 A detailed habitat management plan is required, which should include appropriate and meaningful objectives and targets to ensure the appropriate compensatory habitats are provided and ensure the effectiveness of compensation measures can be appropriately monitored and remedial measures taken if needed. This should include targets in relation to prey populations and marsh harrier activity as targets purely in relation to vegetation type will not demonstrate functionality.

3.487 We also recommend that a monitoring programme is developed to ensure that the compensation area functions as required to avoid effects on the marsh harrier population. Alongside monitoring of the usage of the area by marsh harriers, we also consider that monitoring of prey populations (small mammal and bird populations) is needed, as the functionality of this area depends on its ability to provide replacement foraging rather than purely on the type of vegetation present. This is particularly important given the adequacy of this provision depends on the area being able to support levels of prey in excess of those found in the areas of wetland habitat from which birds will be displaced. Prey monitoring would therefore enable the identification of any need for further habitat creation.

Overall conclusion in relation to requirements of EN-6

3.488 Paragraph 3.5.2 of the Shadow HRA Report Vol. 4: Compensatory Measures[270]states that:

> *"In summary, the analysis presented in Table 1.3 demonstrates that the proposed compensatory measures would meet the requirements of EN-6."*

3.489 Based on our assessment against the requirements of EN-6 and set out in Table 1.3 (as well as the additional criteria contained in the EC's 2018 Managing Natura 2000 guidance[271]), we do not agree that the compensation package is acceptable and meets these requirements.

Impacts of Increased Recreational Pressure on Protected Sites and Species

Background

3.490 Our main concerns in relation to increased recreational pressure on protected sites and species concern both the construction and operation phases of the development and the potential impacts on the Minsmere-Walberswick SPA, SAC and Ramsar site (designated sites) and Sandlings SPA in particular. During the construction phase of the development it is expected that recreational users who may previously have used the Sizewell area and its surroundings may be

[270] Shadow HRA Report Volume 4: Compensatory Measures (APP-152)

[271] European Commission (2018) Managing Natura 2000 sites. The provisions of Article 6 of the 'Habitats' Directive 92/43/EEC

displaced directly by construction activity and associated footpath closures and diversions[272] or indirectly by the perception of loss of tranquillity.

3.491 In addition, the new accommodation campus for workers during the construction phase brings around 2,400 new residents close to the Minsmere-Walberswick designated sites and the Sandlings SPA, part of a total of 7,900 new residents in the wider area during the construction phase. While the effects of the accommodation campus are expected only to last during the construction phase, it is worth remembering the length of that construction period and the effects of recreational displacement of existing users could extend into the operational phase as a result of the formation of new habits and preferences[273].

3.492 Recreational activity may result in disturbance and/or displacement of birds, with the level and type of sensitivity varying by bird species and by season. Ground nesting birds on heaths and beaches are particularly vulnerable to disturbance by people and dogs through potential to cause damage through trampling or predation of nests. Sensitive habitats, such as heathland and vegetated shingle, are also vulnerable to impacts including trampling, nutrient enrichment by dog faeces and fire.

3.493 The Applicant's assessment of recreational changes is set out in ES Vol. 2 Ch. 15 Amenity and Recreation[274] while the ecological implications of increased recreational pressure are assessed in ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology[275] and associated appendices and the Shadow HRA Report Vol. 1, Part 1[276] and Part 4 Appendix 5.10E Recreational Disturbance Assessment[277]. In addition, the Shadow HRA Report Addendum[278] and Vol. 1 ES Addendum Ch. 2 Main Development Site[279] discuss the implications of the Changes to the Application on recreational pressure.

3.494 The National Trust and the RSPB have commissioned Footprint Ecology to review the evidence submitted as part of the Application on recreation issues. Footprint Ecology have therefore provided a report on the sensitivity of designated sites and their features to recreation impacts and the potential for suitable mitigation and a review of the specific ecological reports included in the Application with the aim of checking the robustness of the Applicant's conclusions. These documents form the basis of our representations at this stage and are submitted in full as Annexes to these Written Representations (See Appendix 2 (which contains both reports) Impacts of recreation related to Sizewell C and implications for European sites (Liley & Saunders, 2020a) and Review of Sizewell C application documents and evidence in relation to recreation impacts (Liley & Saunders, 2020b)). We respectfully request that the Examining Authority review these reports in full as the text above and below are only summaries of the points covered within those Reports.

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[272] See Figures 15L1-15L3 of Volume 2 Main Development Site Chapter 15 Amenity and Recreation Appendices Appendix 15I Rights of Way and Access Strategy (APP-270)

[273] Predicted changes to recreational use arising from additional residents (construction workers) and displaced visitors are summarised in Impacts of recreation related to Sizewell C and implications for European sites (Liley & Saunders, 2020a), Maps 2 and 3 in Appendix 2

[274] ES Vol. 2 Main Development Site Ch. 15 Amenity and Recreation (APP-267)

[275] ES Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology (AS-033)

[276] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 (APP-145)

[277] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 4 Appendix 5.10E Recreational Disturbance Assessment (APP-148)

[278] Shadow HRA Report Addendum (AS-173)

[279] Vol. 1 ES Addendum Ch. 2 Main Development Site (AS-181)

Sites and species of concern

3.495 The Minsmere-Walberswick SPA[280] was classified in 1991 and contains extensive areas of freshwater and coastal grazing marsh, coastal reedbeds, saltmarsh, lowland heathland, woodland, intertidal mud and mixed sediment. It supports the following qualifying features:

- Avocet (*Recurvirostra avosetta*), Breeding
- Bittern (*Botaurus stellaris*), Breeding
- Gadwall (*Mareca strepera*), Breeding
- Gadwall (*Mareca strepera*), Non-breeding
- Greater white-fronted goose (*Anser albifrons albifrons*), Non-breeding
- Hen harrier (*Circus cyaneus*), Non-breeding
- Little tern (*Sternula albifrons*), Breeding
- Marsh harrier (*Circus aeruginosus*), Breeding
- Nightjar (*Caprimulgus europaeus*), Breeding
- Shoveler (*Spatula clypeata*), Breeding
- Shoveler (*Spatula clypeata*), Non-breeding
- Teal (*Anas crecca*), Breeding

3.496 The Sandlings SPA[281] was classified in 2001. It comprises acid grassland and heather heathland on sandy soils and supports the following qualifying features:

- European nightjar (*Caprimulgus europaeus*), Breeding
- Woodlark (*Lullula arborea*), Breeding

3.497 Natural England has set site conservation objectives for the SPAs[282] as follows:

> *"The objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:*
>
> - *the extent and distribution of the habitats of the qualifying features*
> - *the structure and function of the habitats of the qualifying features*
> - *the supporting processes on which the habitats of the qualifying features rely*
> - *the populations of each of the qualifying features*
> - *the distribution of qualifying features within the site"*

3.498 In addition, the Sandlings SPA and Minsmere-Walberswick SPA and surrounding suitable habitats support important populations of breeding stone curlew and breeding Dartford warbler. Both of these species are particularly vulnerable to recreational pressure, with stone curlew known to be very sensitive to human disturbance. The population of stone curlew on the Suffolk coast is around 16 pairs based on the five year mean from 2016 to 2020 while that of Dartford warbler is around 130 pairs for the same period (Mel Kemp[283], pers. comm.).

3.499 The Minsmere-Walberswick Ramsar site[284] was designated in 1976 under

- Ramsar criterion 1 for containing a mosaic of marine, freshwater and marshland with a rare transition in grazing marsh ditch plants from brackish to fresh water and

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280 Minsmere-Walberswick SPA: http://publications.naturalengland.org.uk/file/5944954236960768

281 Sandlings SPA: http://publications.naturalengland.org.uk/file/6019964297281536

282 Minsmere-Walberswick SPA COs: http://publications.naturalengland.org.uk/file/4811128974868480 and Sandlings SPA COs: http://publications.naturalengland.org.uk/file/6246575764668416

283 RSPB Minsmere Warden

284 Minsmere-Walberswick Ramsar: http://jncc.gov.uk/jncc-assets/RIS/UK11044.pdf

- Ramsar criterion 2 for supporting nationally scarce plants, British Red Data Book invertebrates and nationally important numbers of breeding and wintering birds (bittern, gadwall, shoveler, teal, marsh harrier, avocet and bearded tit).

3.500 As the breeding assemblage species designated under Ramsar criterion 2 are also SPA features (with the exception of bearded tit, which as a reedbed species is unlikely to be significantly affected by changing recreational use), our comments regarding breeding marsh harrier, bittern, gadwall, teal and shoveler of the Minsmere-Walberswick SPA also apply to the Minsmere-Walberswick Ramsar site.

3.501 The Minsmere to Walberswick Heaths and Marshes SAC[285] was designated in 2005 for its annual vegetation of drift lines, perennial vegetation of stony banks and European dry heaths. Natural England has set site conservation objectives[286] for the SAC as follows:

"Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the Favourable Conservation Status of its Qualifying Features, by maintaining or restoring;

- *the extent and distribution of qualifying natural habitats and habitats*
- *the structure and function (including typical species) of qualifying natural habitats, and*
- *the supporting processes on which qualifying natural habitats rely"*

Summary of concerns relating to increased recreational pressure

3.502 The following is taken from the Summary section of the Footprint Ecology report found in Appendix 2, Impacts of recreation related to Sizewell C and implications for European sites (Liley & Saunders, 2020a). This report sets out Footprint Ecology's evidence review of the impacts of increased recreational pressure and explains our concerns regarding the likely impacts on designated sites arising from the Sizewell C proposals. Please see Appendix 2 for detailed comments.

"The area around Sizewell is of outstanding importance for nature conservation and contains a mix of habitats and species that are unique in the UK. The area is also popular for recreation, which includes tourists, day visitors, and local residents. There is a difficult balancing act to ensure access and conservation do not conflict in a relatively small area.

The Sizewell C proposal would result in changes in recreation use as it would involve a large construction workforce that would be living and working in the area. In addition, construction work would displace existing recreation use (for example through footpath diversions, noise, traffic, etc). We review the impacts of recreation on the nature conservation interest of relevant sites, focussing on the European sites (i.e. those that are subject to particularly strict protection) and their qualifying features.

We highlight a wide range of vulnerable features and risks to their conservation interest resulting from recreation use. Likely significant effects are clear for a range of sites and interest features and, in particular, we highlight:

- *Annual vegetation of drift lines (Minsmere to Walberswick Heaths & Marshes SAC), risks from damage and contamination;*
- *Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC), risks from damage and contamination;*

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285 Minsmere to Walberswick Heaths and Marshes SAC

286 Minsmere to Walberswick Heaths and Marshes SAC Site Conservation Objectives

- *European dry heaths (Minsmere to Walberswick Heaths & Marshes SAC), risks from fire and contamination;*
- *Wintering waterbirds, including White-fronted Goose (Minsmere to Walberswick SPA), risks from disturbance;*
- *Breeding birds, including Nightjar, Avocet, and Little Tern (Minsmere to Walberswick SPA), risks from disturbance;*
- *Breeding Nightjar (Minsmere to Walberswick SPA), risks from fire, and;*
- *Breeding Woodlark and Nightjar (Sandlings SPA), risks from disturbance and fire.*

People coming to the area to live and work in the vicinity will seek nearby greenspace for recreational use that is not necessarily related or connected to the area's conservation interest, and such use will include exercise and dog walking. It may extend to sea angling, barbeques, swimming, and beach recreation. Such use is likely to be away from areas perceived as 'nature reserves' (i.e. where there are hides, wardens, and a fee to enter) and instead focus their use at more peripheral locations. These are often still part of the European site and do not have necessary infrastructure to manage access. We would expect construction workers based in the area to utilise more informal parking and locations where they can easily access beaches, quiet areas of heath, etc. Use by construction workers is also likely to take place in the early mornings, evenings, and around shift work and therefore not necessarily fit with peak visitor use and the current visitor profile.

We provide an overview of the mitigation that is likely to be necessary, highlighting that a package of measures will be required, and this will need to include monitoring to help pick up emerging issues."

Summary of concerns relating to the assessment of recreational pressure

3.503 The following is taken from the Summary section of the Footprint Ecology report found in Appendix 2, Review of Sizewell C application documents and evidence in relation to recreation impacts (Liley & Saunders, 2020b). This report sets out Footprint Ecology's review of the Applicant's assessment of increased recreational pressure arising from the Sizewell C proposals and explains our concerns regarding the adequacy of the assessment and the mitigation and monitoring proposed thus far. Please see Appendix 2 for detailed comments.

"Footprint Ecology have been commissioned by the National Trust and RSPB to provide a review of evidence (submitted as part of the Sizewell C application) on recreation issues. The construction of the power station will require a peak workforce of 7,900 construction workers, leading to a marked increase in the number of residents in the area. In addition, construction will result in the closure of access routes and the displacement of visitors due to path closures, noise, additional traffic, etc. Marked changes in local access are therefore likely. The Sizewell C site is directly adjacent to multiple European sites where there are existing rights of access and numerous species and habitats vulnerable to recreation impacts.

Recreation impacts include disturbance, trampling, increased fire risk, and contamination (e.g. spread of invasive species, dog fouling, etc). Some of the key sensitive areas are already carefully managed, with resources in place to both manage access and allow visitors to experience the area while minimising harm. Areas such as the main car park at Minsmere draw visitors from across the country and are well-known visitor destinations, drawing visitors for the wildlife spectacle and nature interest. Here there are well marked paths, hides, screens, wardens and careful management to allow wildlife to flourish and people to be able to experience it.

Concern with respect to the Sizewell C proposals and recreation impacts therefore relates to vulnerable features within the more peripheral areas of the European sites, where public footpaths, informal parking, and open access provide opportunities for recreational use, and where any issues arising from access are much harder to manage. Access to these locations is more likely to be for dog walking, jogging, picnics, barbeques, fishing, etc. (rather than to see wildlife).

Our review highlights that:

The baseline for current visitor use is inadequate. Count data (visitor numbers) are only provided for a small number of locations (surveyed in 2014) and just for August and November. Many of the smaller, more vulnerable, locations (from an ecological perspective) have not been surveyed.

Predictions of increased use are confusingly presented, with different estimates given in different reports alongside some clear errors. Estimates appear very low compared to national data and are much lower than we would expect, based on our experience of visitor surveys and recreational use of the countryside.

Impacts are poorly assessed, and the consideration of impacts for the following are particularly poor and inadequate:

- *Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC);*
- *European Dry Heaths (Minsmere to Walberswick Heaths & Marshes SAC);*
- *Little Tern (Minsmere-Walberswick SPA);*
- *Nightjar (Minsmere-Walberswick SPA);*
- *Hen Harrier (Minsmere-Walberswick SPA);*
- *Wintering waterbirds (Shoveler, Gadwall, White-fronted Goose) (Minsmere-Walberswick SPA);*
- *Breeding waterbirds (Shoveler, Gadwall, Teal) (Minsmere-Walberswick SPA), and;*
- *Woodlark (Sandlings SPA).*

Mitigation measures are scant and fail to address the nature conservation issues that are likely to arise.

Ultimately the Suffolk Coast is of unique importance for nature conservation and supports a diversity of habitats and species that is unparalleled in the UK, particularly in the area between Slaughden and Southwold. The area also draws high numbers of visitors, resulting in a difficult balancing act to manage both access and nature conservation; it is a crowded area.

Sizewell C would disrupt the balance and the application documents do not provide the robust assessment and mitigation package that we expected. The assessment of impacts is inadequate, and predictions of changes in visitor use are full of inaccuracies, confusing, and poorly put together. Necessary mitigation, targeted to the vulnerable locations, is virtually non-existent. Ultimately, the information as presented in the Shadow HRA does not allow a conclusion of no adverse effect on integrity, from recreation impacts, to be drawn for a range of different species and habitats across multiple European sites."

3.504 This document also sets out suggested points that should be clarified by the Applicant. These can be found in section 5 of the report and are listed below:

"1. There are errors in the calculations of visitor estimates and discrepancies between the different reports. These need to be explained and clear analysis of the recreation impact issues

and necessary mitigation presented. Without this information it is not possible to rule out adverse effects on integrity.

2. There is a lack of detail on assessment of ecological impacts. For example Little Tern, Hen Harrier, and trampling of shingle seem to be poorly addressed. For some of the coastal features there is little indication of how mobile species of dynamic habitats can be protected from disturbance. We would have expected to see much more detailed mapping of visitor flows and numbers (both current and predicted), providing heat maps of footfall. These maps would then be overlaid with ecological data and checked with site managers/local organisations287.

3. Further information is required concerning the coast path diversion. How often will the diversion of visitors to Eastbridge be in place? How will parking issues be addressed in Eastbridge and how will the diverted path cope with increased use, given that it is prone to waterlogging in the winter?

4. There is a lack of mitigation to resolve recreation impacts at the following locations[288]:

- *Eastbridge area (including path to coast);*
- *Periphery of Minsmere Reserve, particularly between Eastbridge and Westleton;*
- *Dunwich Heath;*
- *Beach at Dunwich, including vegetated shingle below the cliffs and marshes to north;*
- *Dunwich Forest;*
- *Walberswick Marshes;*
- *Beach area at Minsmere and south towards Sizewell;*
- *Beach area between Thorpeness and Aldeburgh (vegetated shingle);*
- *North Warren, and;*
- *Aldringham Walks.*

5. As part of a robust mitigation package we would expect to see provision for an increased wardening presence, public education and awareness raising (including a focus on dog fouling and dogs running off-lead), protocols to resolve fire risks, etc. These seem to be alluded to in the Planning Statement but nowhere else.

6. There is no comprehensive monitoring programme to address visitor numbers and use. This needs to be drawn up and integrated into mitigation delivery."

3.505 We request that the Applicant submits further information to the Examination to address these points.

Comments on implications of the Changes to the Application on recreational pressure

3.506 The two reports referenced above were commissioned before the proposals for Changes to the Application were made. The below comments therefore explain how the changes to the proposals affect our concerns around recreational pressure.

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287 We understand some checks were made early on in the assessment with relevant organisations, but the expertise of site managers in checking model predictions and pin-pointing where there are concerns with recreation impacts and how they should be addressed has been missed entirely.

288 See Figures 001 and 002 of Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 4 Appendix 5.10E Recreational Disturbance Assessment (APP-148)

Estimated additional visits to Minsmere and Dunwich Heath

3.507 Paragraphs 8.3.5–10 of the Shadow HRA Report Addendum[289] discusses the uplift in visitor numbers at key locations obtained when those participants in the visitor surveys who stated they would be displaced but not where they would be displaced to are attributed to the main preferred alternative sites. A significant uplift in visits to Eastbridge (from an increase of 20.1% to 31.8%) and Minsmere Outer (from an increase of 50.3% to 79.3%) is described, however, despite reference to Dunwich Heath no figures are provided for this site. It is stated that the two sites of relevance to the HRA are Eastbridge and Minsmere Outer but given that Dunwich Heath forms part of the Minsmere-Walberswick SPA those figures should also be provided here. We have also not been able to find these figures elsewhere in the updated documents. Also, despite the assertions that mitigation for breeding heathland birds has already been proposed, at this stage details of the mitigation and monitoring (of the effectiveness of that mitigation and to check potential effects are as predicted) plan are not available, hence we cannot agree with the conclusions that no impacts are likely. The significance of the uplift at the sites quoted (and assumed at Dunwich Heath in the absence of figures) increases our concern and increases the need for robust mitigation, including additional wardening.

Changes affecting beach users

3.508 The Changes to the Application include moving the marine tunnelling works outside the cut-off wall and the added construction of the new temporary BLF[290]. Vol. 1 ES Addendum Ch. 2 Main Development Site[291] section 2.10 acknowledges additional adverse effects on noise and views experienced by beach users as a result of these changes but argues that they are of limited significance as impacts are already *major adverse*. However, we are concerned that the potential for additional displacement particularly affecting the beach frontage of the Minsmere-Walberswick designated sites is not assessed.

3.509 Paragraph 2.2.67 of Vol. 1 ES Addendum Ch. 2 Main Development Site also explains that the Suffolk Coast Path would be redirected up and down the shoreline as necessary to facilitate construction of the enhanced permanent BLF. We consider that the potential for increased impacts on vegetated shingle of the Minsmere-Walberswick SAC should therefore also be assessed.

In-combination assessment

3.510 Vol. 3 Ch. 10 Inter-relationships, Project-wide, Cumulative and Transboundary Effects Appendix C (AS-201)[292], provides an assessment of cumulative terrestrial ecology and ornithology effects with the onshore cable route construction of the Scottish Power Renewables (SPR) offshore wind projects. This notes the potential for in-combination effects of direct land take and disturbance from the SPR projects along with recreational pressure and noise and visual disturbance from Sizewell C construction. It concludes (regarding the area of direct effect) that as the 0.7km SPA block "only" supports approximately 3% of the SPA breeding nightjar population and approximately 9% of the SPA breeding woodlark population that, with the mitigation proposed by both developers, effects would not be significant. This assessment does not consider the potential for displacement due to the construction of Sizewell C to increase/change the predicted impacts of the SPR projects and we therefore consider that this assessment requires updating.

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289 Shadow HRA Report Addendum (AS-173)

290 See photowire visualisations for zones C16 and C20 in Figure 2.8.20 of Volume 2 Environmental Statement Addendum Chapter 2 Main Development Site Figures Part 4 of 4 (AS-193)

291 Vol. 1 ES Addendum Ch. 2 Main Development Site (AS-181)

292 Vol. 3 Ch. 10 Inter-relationships, Project-wide, Cumulative and Transboundary Effects Appendix C (AS-201)

Conclusions regarding impacts of recreational pressure on protected sites and species

3.511 In conclusion, based on the above concerns and the evidence provided at this stage, we do not agree that adverse *effects on integrity* can be ruled out for the following sites and features for impacts from the project alone:

- Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC);
- European Dry Heaths (Minsmere to Walberswick Heaths & Marshes SAC);
- Little Tern (Minsmere-Walberswick SPA);
- Nightjar (Minsmere-Walberswick SPA);
- Hen Harrier (Minsmere-Walberswick SPA);
- Wintering waterbirds (Shoveler, Gadwall, White-fronted Goose) (Minsmere-Walberswick SPA);
- Breeding waterbirds (Shoveler, Gadwall, Teal) (Minsmere-Walberswick SPA);
- Woodlark (Sandlings SPA), and;
- Nightjar (Sandlings SPA).

3.512 In relation to this Application in combination with other projects, we do not agree that *adverse effects on integrity* can be ruled out for the following sites and features:

- Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC);
- European Dry Heaths (Minsmere to Walberswick Heaths & Marshes SAC);
- Little Tern (Minsmere-Walberswick SPA);
- Nightjar (Minsmere-Walberswick SPA);
- Hen Harrier (Minsmere-Walberswick SPA);
- Wintering waterbirds (Shoveler, Gadwall, White-fronted Goose) (Minsmere-Walberswick SPA);
- Breeding waterbirds (Shoveler, Gadwall, Teal) (Minsmere-Walberswick SPA);
- Woodlark (Sandlings SPA), and;
- Nightjar (Sandlings SPA).

Mitigation and monitoring

3.513 One of our key concerns described above and in the detailed reports by Footprint Ecology is the lack of detail around the proposals for mitigation and monitoring of impacts. The Application documents variously refer to monitoring and mitigation of recreational pressure on designated sites through the Rights of Way and Access Strategy[293] and through a monitoring and mitigation plan. However, no detailed monitoring or mitigation proposals have been presented for the Minsmere-Walberswick designated sites or the Sandlings SPA and therefore, as discussed above, we cannot agree that this is a sufficient basis on which to rule out *adverse effects on integrity*.

3.514 We note that there has been some progress with mitigation for the effects of the worker campus with a proposed contribution to the Suffolk Councils' Recreation Avoidance and Mitigation Strategy contained within Schedule 11 of the most recent iteration of the Draft Section 106 Agreement[294]. As discussed further in Appendix 2, we consider that a robust and detailed plan for monitoring and mitigation of recreational impacts on designated sites from recreational displacement must also be developed along with provision of suitable alternative natural green

[293] Volume 2 Main Development Site Chapter 15 Amenity and Recreation Appendices Appendix 15I Rights of Way and Access Strategy (APP-270)

[294] Draft s106 Agreement (REP1-007)

space for additional residents during the construction period. We consider that such provisions should be secured through requirements in the DCO**.**

3.515 We welcome the continued engagement of the Applicant with us and others in discussions on this topic. We understand that the Applicant intends to submit mitigation and monitoring plans for the relevant protected sites at a forthcoming deadline. We look forward to the opportunity to update our position on the impacts of recreational pressure on protected sites and species following submission and review of these plans.

Marine ecology - sites and species of concern

3.516 Our concerns primarily relate to potential impacts on

- **non-breeding red-throated diver, foraging common and little terns (during the breeding season) of the Outer Thames Estuary SPA,**
- **breeding little terns of the Minsmere-Walberswick SPA and**
- **breeding Sandwich terns of the Alde-Ore Estuary SPA[295].**

3.517 Natural England has set site conservation objectives[296] for all three of the SPAs as follows:

> *"Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;*
>
> - *The extent and distribution of the habitats of the qualifying features,*
> - *The structure and function of the habitats of the qualifying features,*
> - *The supporting processes on which the habitats of the qualifying features rely,*
> - *The population of each of the qualifying features, and,*
> - *The distribution of the qualifying features within the site."*

3.518 The population of little terns at **the Minsmere-Walberswick SPA[297]** at the time of designation in 1991 was 32 breeding pairs. Since then Natural England's **Supplementary Advice on Conservation Objectives[298]** for the SPA notes that there has been a significant decline with a five year mean peak count of 1.6 breeding pairs from 2014 to 2018. It is notable that the little tern population of the SPA varies significantly year on year with 10 pairs noted at RSPB Minsmere in 2019 but no breeding at RPSB Minsmere prior to that since 2009 (R. Harvey[299], pers. comm.). The Supplementary Advice on Conservation Objectives note that this may be because the little tern population of Norfolk and Suffolk behave as a metapopulation with birds moving between sites year on year. However, given the general decline in numbers at the Minsmere-Walberswick SPA, a restoration target has been set for this species to over 32 breeding pairs.

3.519 The little tern population of the Minsmere-Walberswick SPA also form part of the designated population of the **Outer Thames Estuary SPA[300]**. This SPA was reclassified in 2017 following changes to the designated site boundary and to add breeding common and little terns to its qualifying features, as this marine SPA provides foraging habitat for these species. The population

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295 See ES Vol. 2 Ch. 14 Terrestrial Ecology and Ornithology Appendix 14A2 - Designated Sites Annex 14A2.1: Figures 14A2.1 - 14A2.3 (APP-228) for locations of designated sites

296 Outer Thames Estuary SPA Site Conservation Objectives, Minsmere-Walberswick SPA Site Conservation Objectives, Alde-Ore Estuary SPA Site Conservation Objectives:

297 Minsmere-Walberswick SPA

298 Minsmere–Walberswick SPA - Supplementary Advice on Conservation Objectives please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

299 RSPB Minsmere Site Manager

300 Outer Thames Estuary SPA

of little terns at the time the SPA was reclassified in 2017 was 746 breeding individuals which are derived from colonies at Great Yarmouth North Denes, Minsmere-Walberswick, Benacre to Easton Bavents, Alde-Ore Estuary, Foulness and Thanet Coast and Sandwich Bay SPAs. The common tern population of the SPA at the time of reclassification was 532 breeding individuals made up of birds from around the south-east of England. The target for both of these species at this SPA has been set as "maintain" the population as insufficient time has elapsed since reclassification to establish trends, however it should be noted that the targets include a need to avoid deterioration from current levels.

3.520 The Outer Thames Estuary SPA is also designated for its non-breeding red-throated diver population. The SPA was originally classified in 2010 on the basis that it supported an internationally important population of 6,466 individual wintering red-throated divers. Subsequent aerial surveys of the site have revealed that it currently supports significantly higher numbers of red-throated divers - the current SPA population is estimated at 18,079 overwintering individuals (peak mean 2012/3 – 2017/18) (Irwin *et al.*, 2019)[301]. Importantly, the Supplementary Advice on Conservation Objectives[302] for this feature set a target to:

> *"Reduce the frequency, duration and / or intensity of disturbance affecting roosting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed"*

3.521 The Supplementary Advice on Conservation Objectives note the vulnerability of red-throated diver to disturbance by boats and the strong stress response exhibited by birds in response to such disturbance from marine activity and construction. The population target for this feature is to maintain the population at over 18,079 individuals, again noting the need to avoid deterioration from current levels.

3.522 The Sandwich tern population of **the Alde-Ore Estuary SPA[303]** appears to have undergone significant decline since the site's designation. The Supplementary Advice on Conservation Objectives[304] for this site note that nesting has been sporadic with a maximum of 15 pairs recorded in 2003. Although a numeric population target has not been set for this species, the target is to restore the population. We acknowledge that the lack of recent breeding data for Sandwich terns at the Alde-Ore Estuary SPA makes it difficult to assess potential impacts of the Application on this species, however, as above with the little tern population, there is a need to ensure that the development does not compromise the ability to restore this population.

3.523 We are also concerned that impacts on the Minsmere-Walberswick Ramsar site[305] require further investigation. The Minsmere-Walberswick Ramsar site was designated in 1976 under Ramsar criterion 1 for containing a mosaic of marine, freshwater and marshland with a rare transition in grazing marsh ditch plants from brackish to fresh water and under criterion 2 for supporting nationally scarce plants, British Red Data Book invertebrates and nationally important numbers of breeding and wintering birds (bittern, gadwall, shoveler, teal, marsh harrier, avocet and bearded tit).

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301 Irwin, C., Scott, M., S., Humphries, G. & Webb, A. (2019) HiDef report to Natural England - Digital video aerial surveys of red-throated diver in the Outer Thames Estuary Special Protection Area 2018. Natural England Commissioned Reports, Number 260.

302 Outer Thames Estuary SPA - Supplementary Advice on Conservation Objectives please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

303 Alde -Ore Estuary SPA

304 Alde-Ore Estuary SPA – Supplementary Advice on Conservation Objectives please note once webpage opened you have to tick which species and then ask it to show *attributes and targets for selected features*.

305 Minsmere-Walberswick Ramsar, Minsmere-Walberswick Ramsar Information Sheet

Indirect impacts on birds from fish mortality arising from the cooling water system

Introduction

3.524 Our concerns around the intake of marine organisms via the cooling water system and the subsequent discharge of dead and moribund biota (fish and other organisms) via the Fish Recovery and Return (**FRR**) system relate to the potential effects on bird predator species arising from reduced prey availability and the effects of changes to water quality as a result of the discharged material.

3.525 We understand that water will be abstracted during the operational phase via two cooling water intakes situated around 3km offshore. The cooling water intakes are proposed to be fitted with Low Velocity Side Entry (**LVSE**) heads with the aim of reducing the abstraction of marine organisms, however, some marine organisms will be impinged on the drum screens protecting the intake heads, while other smaller organisms will be entrained into the power station cooling water system. Larger organisms impinged on the drum screens would be removed and discharged via the FRR system, and while some organisms may survive, others would suffer high mortality rates due to pressure or mechanical stress. Those organisms entrained into the cooling water system would be exposed to thermal uplift, increased pressure, hydrazine and chlorination and are likely to experience high levels of mortality.

Impingement and entrainment

3.526 We are concerned about the potential impacts of impingement and entrainment on important prey species for birds from the SPA populations. Ecologically important species present in the Greater Sizewell Bay include sprat, herring, anchovy, whiting, seabass, Dover sole, gobies and dab. Paragraph 8.10.63 of the Shadow HRA Report[306] also notes that the diet of non-breeding red-throated diver in the North Sea includes clupeids (herring and sprat), gadoids (including whiting and cod), gobies, sandeels and smelt.

Thresholds for assessing the effects of impingement and entrainment

3.527 Paragraphs 8.3.78 – 8.3.86 of the Shadow HRA Report[307] explain the approach to the assessment of significance of the effects of impingement and entrainment on bird prey populations. This is based on the use of a percentage of the Spawning Stock Biomass (**SSB**) of each fish species as an indicative threshold for significance, with 1% of the SSB used for commercially exploited stocks and conservation species (including stocks that are currently unexploited) and 10% of the SSB used for other unexploited stocks. The scale of the SSB for most species will be large – paragraph 8.3.84 quotes the examples of cod, whiting, herring and sprat, which are all based on the North Sea or a still larger area. It is acknowledged that local effects on fish populations may occur, for example entrainment and impingement would be likely to affect prey populations within the Greater Sizewell Bay[308], and that these could have significant impacts on the SPA birds, particularly during the breeding season when birds are central place foragers tied to foraging within a certain radius of their nest site. Such local impacts however, may not be detected through the use of thresholds based on the SSB and we therefore do not agree that these are suitable for use in the HRA to assess the significance of changes to prey availability for birds from SPA populations.

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306 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

307 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

308 See Figure 1 on p16 of ES Vol. 2 Ch. 20 Coastal Geomorphology and Hydrodynamics Appendix 20A Coastal Geomorphology and Hydrodynamics: Synthesis for Environmental Impact Assessment (APP-312)

3.528 We therefore welcome the updated assessment provided in section 8 of Report TR406[309] which includes consideration of fish depletion at the scale of Greater Sizewell Bay. This explains that:

> *"Local scale depletion due to Sizewell C and Sizewell B acting in-combination is predicted to be less than 2.5% for demersal species[310], 1.5% for pelagic species[311] and less than 1% of epibenthic species[312] during periods of seasonal presence… assuming 10% daily replenishment of fish from the wider area… Even at exchange rates as low as 5% per day local depletion of pelagic species is expected to be in the order of 3% with the abundance of epibenthic species reduce by less than 2% and demersal species by less than 5% with both stations operating."*

3.529 It is then stated that, as these levels are smaller than the interannual variability in the populations of these species, the effects will be insignificant. We disagree with this statement as low levels of fish availability (even within the bounds of natural variability) are known to affect the breeding success of tern populations. We also request that an assessment is provided of how the predicted local scale depletions are likely to affect fish populations in Greater Sizewell Bay over the time scale of the operation of the power plant, as the current assessment does not appear to consider the year on year (cumulative) nature of such impacts. This constant removal of fish year every year could lead to increasing failures of e.g. tern breeding colonies and compromise the ability to meet SPA conservation objectives to restore or maintain such populations.

Use of Equivalent Adult Values (EAVs)

3.530 The projected losses from entrainment for the Application are set out in Tables 22.105 (unadjusted for survival and Equivalent Adult Values – EAV) and 22.106 (adjusted for equivalent spawning females or EAVs) of ES Ch. 22 Marine Ecology and Fisheries[313]. The unadjusted values for Sizewell B are also presented in Table 22.104 and from this it can be seen that the Application is predicted to entrain roughly 2½ times as many individuals of each species as Sizewell B. Although we understand that entrained fish are usually at the egg, larvae and juvenile stage and that many of these would not survive to adulthood even if they were not entrained, we do have some concerns around the use of EAVs in the assessment of impacts on predatory birds.

3.531 Paragraph 22.8.503 explains the approach of calculating EAVs for eggs, larvae and juvenile fish entrained, based on the likely proportion surviving to adulthood, for the purposes of assessing commercial impact. However, it is likely that juveniles of some species are taken by predatory birds, particularly as e.g. little terns are known to feed smaller prey to young chicks (Bogliani *et al.*, 1994[314], Paiva *et al.*, 2006[315]) and hence EAVs could underestimate ecological impacts. We therefore recommend that the unadjusted values for juvenile fish are considered within the assessment of indirect impacts on birds. We also support the Environment Agency's concerns[316] that EAVs do not account for the spawning potential of fish and hence under-estimate mortality, and therefore support the need for consideration of Spawner Production Foregone (**SPF**) figures in this assessment.

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309 Cefas BEEMS report TR406 in Vol. 3 Ch. 2 ES Addendum Marine Ecology and Fisheries Appendix 2.17.A (AS-238)

310 Which includes sea bass, cod, whiting and grey mullet

311 Which includes sprat, herring, anchovy and shad

312 Which includes eel, lampreys, sole and sand goby

313 ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)

314 Bogliani, G., Fasola, M., Canova, L., and Saino, N. (1994) Prey selection by parents and chicks of the little tern *Sterna albifrons*. Avocetta 18: 9-11

315 Paiva, V., Ramos, J., Catry, T., Pedro, P., Medeiros, R. and Palma, G. (2006) Influence of environmental factors and energetic value of food on Little Tern *Sterna albifrons* chick growth and food delivery. Bird Study 53: 1-11

316 p15 of the Environment Agency's Relevant Representation (RR-0373)

Impingement and fish nursery grounds

3.532 Tables 22.111 and 22.112 of ES Ch. 22 Marine Ecology and Fisheries[317] show the predictions of impingement mortality for Sizewell C without mitigation and with mitigation (**LVSE and FRR**) respectively. These show that the species with the highest predicted levels of impingement are those that are known to be ecologically important, including sprat, herring, whiting, seabass, sand goby, sole, dab and anchovy[318]. These species are important prey species for SPA bird populations, with sprat, herring and whiting in particular noted in the diet of red-throated diver. Paragraph 22.4.60 also notes that of these, sprat, herring, whiting and seabass have nursery grounds within the Greater Sizewell Bay and that Dover sole has both spawning and nursery grounds in this area.

3.533 Section 8 of Report TR406[319] (p101) also acknowledges the potential for localised effects if fish depletion affects nursery grounds, although it is concluded that nursery grounds for the species found in Greater Sizewell Bay are widely distributed and hence of low importance. We are however concerned that impingement affecting these nursery grounds could affect local prey availability for predatory birds, particularly those with restricted foraging ranges, such as little tern.

Impingement of eels

3.534 We note that Tables 22.111 and 22.112 of ES Ch. 22 Marine Ecology and Fisheries[320] also indicate that eels are likely to be impinged by the Application. This assessment is expanded in the updated information in section 6.6.2 of Report TR406[321] and impacts on the European eel stock are concluded to be negligible. However, we note that in the absence of mitigation (or if mitigation proved ineffective), combined impingement and entrainment mortality exceeds 1% SSB[322] for European eel. Therefore, given that eels form an important food source for bittern of the Minsmere-Walberswick SPA, impacts on local stocks should be assessed in the Shadow HRA.

Assessment of entrainment of sand gobies323

3.535 Paragraph 22.8.516 of ES Ch. 22 Marine Ecology and Fisheries[324] states that:

> *"the estimated number of sand gobies that would be entrained by Sizewell C represents approximately 1.4% of the abundance".*

3.536 As sand gobies are not a commercially exploited species, a threshold of 10% of the SSB is applied to indicate significant effects and it is therefore concluded that this level of effect will not be significant. However, using the criteria for recommended thresholds set out in paragraph 8.3.78 of the Shadow HRA Report[325], a threshold of 1% should be applied where a species is of ecological importance (defined within ES Ch. 22 Marine Ecology and Fisheries as being common and/or abundant enough to play a key trophic role within the ecosystem[326]). We argue that as sand goby are an ecologically important species (as they are prey species to red-throated diver, little tern and common tern of the Outer Thames Estuary SPA, little tern of the Minsmere-Walberswick SPA

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317 ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)
318 Of which sea bass and whiting are demersal species, sprat, herring and anchovy are pelagic and sand goby, sole and dab are epibenthic.
319 Cefas BEEMS report TR406 in Vol. 3 Ch. 2 ES Addendum Marine Ecology and Fisheries Appendix 2.17.A (AS-238)
320 ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)
321 Cefas BEEMS report TR406 in Vol. 3 Ch. 2 ES Addendum Marine Ecology and Fisheries Appendix 2.17.A (AS-238)
322 Spawning Stock Biomass
323 An epibenthic species
324 ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)
325 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
326 Or a key role in the food chain

and Sandwich terns of the Alde-Ore Estuary SPA), the 1% threshold should be applied and this impact considered potentially significant.

Entrainment and climate change

3.537 Paragraphs 22.6.242 – 22.6.247 explain that mortality due to temperature shock for the egg and larval life stages of many fish species increases rapidly once maximum temperatures exceed 30°C. Climate change will result in an increasing number of days per year where entrainment temperatures (ambient +11.6°C) exceed 30°C; 57 days per year by 2030 and 100 days per year by 2055. It is acknowledged that this would lead to higher entrainment mortality rates and we are concerned that this could affect species of ecological importance and potentially combine with other climate pressures to negatively affect SPA seabirds (which has not been considered).

Total mortality from impingement and entrainment

3.538 Sprat and herring are ecologically important species likely to be present in the diet of red-throated diver, little tern and common tern of the Outer Thames Estuary SPA, little tern of the Minsmere-Walberswick SPA and Sandwich terns of the Alde-Ore Estuary SPA and are the species predicted to be most affected by impingement and entrainment from the Application. Paragraph 8.3.97 of the Shadow HRA Report[327] states that:

> *"for sprat the combined estimated annual mortality for entrainment and impingement at Sizewell C (with the low velocity side entry and FRR[328] in place) was 2,249,905 equivalent adult values"*
>
> while:
>
> *"for herring the estimated combined annual mortality was 724,095 equivalent adult values".*

3.539 The paragraph goes on to conclude that each of these impacts are lower than the 1% threshold of the relevant SSBs. This appears to be a significant level of fish mortality and again we raise our concerns that assessing this at the level of SSBs may miss local impacts on Greater Sizewell Bay which will be more relevant to the assessments of impacts on bird predators required for the purposes of HRA. We also note our concerns with the use of EAVs[329] which may lead to underestimation of ecological effects, as discussed above.

3.540 Based on section 6.7 of Report TR406[330] we note that in the absence of mitigation (or if mitigation proved ineffective), combined impingement and entrainment mortality exceeds 1% SSB for sea bass, European eel, thin-lipped grey mullet and sand goby. As discussed below, we are concerned that the mitigation proposed does not accord with industry best practice (Turnpenny *et al.* 2010)[331] and therefore are concerned that impacts on these species could be significant (also noting our concerns about the use of 1% SSB as a threshold for significance).

3.541 The Shadow HRA goes on to report conclusions for the indirect impacts of impingement and entrainment on predator species including red-throated diver (8.10.63-65), little tern (8.10.9) and common tern (8.10.44-45) of the Outer Thames Estuary SPA, little tern of the Minsmere-Walberswick SPA (8.8.348-350) and Sandwich terns of the Alde-Ore Estuary SPA (8.3.99-100). Conclusions that the mortality of each fish species equates to less than 1% of the SSB (or less than

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327 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

328 The Fish Recovery and Return

329 Equivalent Adult Values

330 Cefas BEEMS report TR406 in Vol. 3 Ch. 2 ES Addendum Marine Ecology and Fisheries Appendix 2.17.A (AS-238)

331 Turnpenny, A.W.H., Coughlan, J., Ng, B., Crews, P., Bamber, R.N., & Rowles, P. (2010) Cooling water options for the new generation of nuclear power stations in the UK. Environment Agency

10% for sand goby – see our comments above) are used to justify a conclusion of no *adverse effects on the integrity* of those SPAs from this impact. We are concerned that these discussions do not consider total prey availability losses or local effects within Greater Sizewell Bay, especially when considered together and the potential cumulative effects of losses year on year.

3.542 Section 8 of Report TR406 updates this assessment through consideration of the effects of local scale depletions in fish populations and the effects on foraging birds of the Minsmere-Walberswick, Outer Thames Estuary and Alde-Ore Estuary SPAs. This acknowledges that the restricted foraging range of little terns makes them particularly vulnerable to the effects of prey depletion. However, we note that we consider this concern relevant to all the SPA tern populations. It explains that little tern numbers at the Minsmere-Walberswick SPA have declined since designation but should recognise the requirement to avoid impacts that could compromise the restoration of this population. The assessment compares the level of local prey depletion with the biomass required to sustain the designated population of little terns. This assessment is inadequate as it does not recognise the impacts of prey depletion on foraging efficiency and success rates. We are concerned that such impacts could reduce their breeding success/productivity as was seen when the construction of Scroby Sands offshore wind farm reduced prey availability for a colony of little terns, as reported in Perrow *et al.* (2011)[332].

Discharge of dead and moribund biota

3.543 Paragraphs 7.4.25- 44 of the Shadow HRA Report[333] explain that the discharge of dead and moribund biota from the FRR system during operation will contribute to biochemical oxygen demand (BOD) and increase nutrient inputs and levels of un-ionised ammonia in the water column. Whilst these impacts would be expected to be fairly local to the outfalls within the Greater Sizewell Bay we are concerned that changes to water quality could combine with other operational impacts on Greater Sizewell Bay, such as the thermal and chemical plumes, to affect prey distribution for SPA bird populations. In paragraph 22.10.29 of ES Ch. 22 Marine Ecology and Fisheries[334], the Applicant states that the discharged material may attract scavenging seabirds in a similar manner to fisheries discards and hence may have beneficial effects on bird populations. It should be noted that red-throated diver (Outer Thames Estuary SPA) and little tern (Outer Thames Estuary SPA and Minsmere-Walberswick SPA) do not forage on discards and hence will not benefit in this way.

3.544 We also note that likely levels of contaminants from the cooling water system in dead and moribund fish are not discussed and the potential for any impacts on gull species which may take dead fish should be considered.

Mitigation

3.545 We are concerned that the design of the cooling water system does not include an Acoustic Fish Deterrent (AFD) or other repulsive technology and therefore that the overall design does not comply with best practice (Turnpenny *et al.* 2010)[335]. Inclusion of an AFD would reduce the potential impacts on fish populations in Greater Sizewell Bay, including species for which nursery or spawning grounds are present and species that are important prey resources for bird features of all the SPAs. Use of an AFD would also have beneficial effects on water quality through reducing the amount of dead and moribund biota discharged to Greater Sizewell Bay. We note that section

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[332] Perrow, M., Gilroy, J., Skeate, E. & Tomlinson, M. (2011) Effects of the construction of Scroby Sands offshore wind farm on the prey base of Little tern *Sternula albifrons* at its most important UK colony. Marine Pollution Bulletin 62: 1661-1670

[333] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

[334] ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)

[335] Turnpenny, A.W.H., Coughlan, J., Ng, B., Crews, P., Bamber, R.N., & Rowles, P. (2010) Cooling water options for the new generation of nuclear power stations in the UK. Environment Agency, p175-176

2.3 of Report TR406[336] states that AFD are too complex to install and maintain in an offshore environment and may not have sufficient longevity. Whilst we defer to the expertise of others (particularly the Environment Agency) regarding this, we request that evidence and case studies around the use of AFDs are presented in order to consider this more fully. We also recommend that figures comparing levels of predicted impingement and entrainment with and without AFD are provided in order to assess its potential efficacy and inform the consideration of their inclusion within the mitigation proposals.

Direct and indirect impacts on birds from the thermal plume

Introduction

3.546 Our concerns around the thermal plume (the warm water discharged from the cooling water system to the sea) during the operational period relate to the potential for indirect impacts on prey species important to bird populations of the Outer Thames Estuary SPA, Minsmere-Walberswick SPA and the Alde-Ore Estuary SPA and the potential for direct effects through displacement of birds loafing or foraging on the water surface. The effects of the thermal plume are assessed on the basis of the uplift in temperature compared to the baseline and the absolute sea surface temperature, with further investigation recommended if uplift exceeds +2°C over the baseline or for absolute temperatures the 98th percentile[337] of less than or equal to 28°C (paragraph 7.4.12 of the Shadow HRA Report[338]).

Increase in absolute sea temperature

3.547 Paragraph 22.6.301 of ES Ch. 22 Marine Ecology and Fisheries[339] explains that:

> *"During the operation of both stations, absolute temperatures of 23°C increase from 89.6ha at the surface… to a worst case of 506.2ha at the surface and 264.4ha at the seabed in 2055."*

3.548 Whilst it goes on to note that if Sizewell B is decommissioned as expected that this extent will reduce significantly after 2055, the increase up to this point is significant and could be of ecological concern. Paragraph 22.8.660 explains that where the thermal plume results in absolute temperatures of less than 27 °C, there are no clear deleterious impacts on species, but in the longer term, that changes in species composition may occur.

3.549 We query whether any such long-term community changes could have effects on predators, e.g. terns foraging in coastal waters in the breeding season. Table 22.123 also indicates the preferred temperatures and upper lethal temperatures for various cold-water fish species, several of which are of importance as prey for SPA bird species. This indicates that temperatures of over 23°C will exceed the preferred temperature for adults of key species such as herring, smelt, dab, whiting and cod and may even exceed the lethal temperature for some species. This could have the effect of displacing these species from the areas affected by the thermal plumes, and the potential effects on prey availability and distribution could result in the short term, in unsuccessful breeding/poor productivity, and in the longer term, displacement of SPA bird species.

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336 Cefas BEEMS report TR406 in Vol. 3 Ch. 2 ES Addendum Marine Ecology and Fisheries Appendix 2.17.A (AS-238)

337 Meaning that, based on a frequency distribution of temperature observations, 98% of observations are less than or equal to 28°C

338 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

339 ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)

Uplift in sea temperature compared to baseline

3.550 Paragraph 8.8.287 of the Shadow HRA Report[340] states that:

> *"Based on the 98th percentile, the 2°C and 3°C thermal plumes for the combined effects of Sizewell B and Sizewell C encompass 61.5% and 41.3% of the foraging range of the Minsmere colony, respectively (Plate 8.12). This compares with overlaps of 48.9% (for the 2°C uplift) and 34.0% (for the 3°C uplift) for the existing thermal plumes that result from Sizewell B. For the Dingle colony foraging range, the equivalent percentage overlaps are 31.9% and 3.3% for the 2°C and 3°C uplifts, respectively, for the combined effects of Sizewell B and Sizewell C, and 11.7% and 0.1% for the 2°C and 3°C uplifts, respectively, for the existing thermal plumes resulting from Sizewell B."*

3.551 This indicates a significant overlap between the foraging area of little terns from the Minsmere-Walberswick SPA (and foraging within the Outer Thames Estuary SPA) and both the +2°C and +3°C thermal plumes and that the extent of the plume increases significantly compared to the baseline with Sizewell B only in operation. It is later stated that little terns currently forage within the Sizewell B +2°C thermal plume, but we note that it is not known whether they will continue to forage in the area encompassed by the +3°C plume. Paragraphs 8.8.292 and 8.10.4 conclude that no *adverse effects on the integrity* of the little terns of the Minsmere-Walberswick SPA or the Outer Thames Estuary SPA exists on the basis that +2°C is an overly precautionary threshold and that the overlap of their foraging range with the +3°C plume is relatively small. On this basis and because common terns from Minsmere and Orfordness only make up a small part of the Outer Thames Estuary SPA population of this species, a conclusion of no *adverse effects on the integrity* is also reached for the common tern feature of the Outer Thames Estuary SPA.

3.552 Regarding red-throated divers of the Outer Thames Estuary SPA, paragraph 8.3.15 states that the overlap of the maximum extent of the predicted 2°C thermal plume at the sea surface for Sizewell C encompasses 4.3% of the area of the Outer Thames Estuary SPA. It is also stated that this part of the SPA holds a relatively small proportion of the red-throated diver population and hence impacts would not be as significant as indicated.

3.553 However, we disagree with this assertion on the basis of the Red-throated Diver Reports for 2012-2013 and 2013-2014 in the Annexes to ES Ch. 14 Appendix 14A7 Part 2[341] indicate the importance of the Greater Sizewell Bay for red-throated diver, with the peak number of red-throated divers recorded across the whole survey area (approximately 2km offshore from Dunwich to Orfordness) during the winter of 2012/13 equating to 19% of the SPA population. These surveys are likely to have underestimated abundance as they were carried out from vantage points (discussed further below) but give an indication of the importance of the area. Mason (Ed.) (2015)[342] also states that red-throated diver densities off the coast of Suffolk are significantly higher north of Orfordness than south of this point. Given that red-throated divers both loaf and forage from the sea surface (rather than diving direct from the air), we are concerned that the thermal plume could result in direct displacement effects on this species in this part of the Outer Thames Estuary SPA (designated for this species originally) and consider that further assessment of the potential for these effects should be provided.

3.554 We also consider that the potential for impacts of the thermal plume to combine with other operational impacts on Greater Sizewell Bay, such as the chemical plumes and other changes to

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340 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

341 ES Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology Appendix 14A7 Ornithology Part 2 of 2 (APP-238)

342 Mason, N. (Ed.) (2015) Suffolk Birds 2014. A review of birds in Suffolk in 2014. Vol. 64. Suffolk Naturalists' Society, Ipswich.

water quality (the cumulative effects), to cause displacement of SPA bird species as a result of direct effects or through effects on prey distribution have not been fully considered.

Direct and indirect impacts on birds from the chemical plumes

Introduction

3.555 The cooling water system will be dosed with chlorine during operation to prevent biofouling and hence maintain the safe operation of the power station. Chlorination is applied seasonally when water temperatures exceed 10°C and occasionally when needed outside this period. As noted by paragraph 22.3.94 of ES Ch. 22 Marine Ecology and Fisheries[343], due to climate change, the period when chlorination is required is expected to lengthen as water temperatures increase (from 219 days per annum in 2030 to 244 days per annum in 2085). This cooling water will be discharged via the cooling water outfall and will contain measurable Total Residual Oxidants (**TRO**) and bromoform (a chlorination by-product) when dosing has been applied.

3.556 Hydrazine will also be added to the cooling water during commissioning (during construction) and on a daily basis (although not continuously) during operation to prevent corrosion. We are concerned that the chemical plumes resulting from the discharge of TROs, bromoform and hydrazine have the potential to affect prey species of birds from the Minsmere-Walberswick, Outer Thames Estuary and Alde-Ore Estuary SPAs and may also have direct displacement or toxic effects on birds.

Effects of the TRO and bromoform plumes on birds and their prey

3.557 Paragraph 22.6.310 of ES Ch. 22 Marine Ecology and Fisheries[344] explains that the Sizewell C TRO plume concentration is expected to exceed the Environmental Quality Standard (EQS) of 10µg/l as a 95th percentile over a sea surface area of 338ha and a seabed area of 2.1ha. For bromoform, the Predicted No-Effect Concentration[345] (**PNEC**) of 5µg/l is exceeded by the Application plume over an area of 52ha at the surface and 0.67ha at the seabed. Both plumes are aligned along the shore and tidally transported north and south of the outfalls. They are distinct from the Sizewell B plumes, which are closer to shore due to the location of the outfalls.

3.558 Our concerns largely relate to potential impacts on fish as prey for bird species of the Outer Thames Estuary, Minsmere-Walberswick and Alde-Ore Estuary SPAs and the potential for fish to be locally displaced from areas affected by the plumes through avoidance behaviour. This potential is acknowledged for both TROs and bromoform in paragraphs 22.8.759 and 22.8.783 respectively of ES Ch. 22 Marine Ecology and Fisheries, although the Shadow HRA Report concludes that impacts of changed prey distribution are not significant due to the limited overlap of the plumes with foraging areas of red-throated diver and common tern of the Outer Thames Estuary SPA and Sandwich terns of the Alde-Ore Estuary SPA and no overlap with foraging areas of little tern of the Outer Thames Estuary and Minsmere-Walberswick SPAs. Whilst we accept that such effects are likely to be localised, we are concerned that effects of these plumes could work in-combination with other potential impacts on prey distribution, for example, from the thermal plume.

3.559 We query the potential for the TRO and bromoform plumes to have direct toxic effects on birds, particularly species such as red-throated diver which loaf and forage from the sea surface and note that no evidence has been provided regarding this. Whilst we understand that chlorination

[343] ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)

[344] ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)

[345] The PNEC can be defined as the concentration of a chemical which marks the limit at which below no adverse effects of exposure in an ecosystem are measured and can be calculated for acute (short term) or chronic (long term) exposure

will be applied seasonally and that red-throated diver are a non-breeding feature of the Outer Thames Estuary SPA, given the number of days per annum that chlorination is likely to required, some overlap would be expected (with red-throated diver typically being present in the area from October to April, although occasionally are seen in other months). Given that potential impact pathways exist, this impact should be considered within the HRA.

3.560 We also query the potential for these plumes to elicit avoidance behaviour from birds and whether the fact that the plumes from Sizewell C will be further offshore than those of Sizewell B could mean that red-throated divers are more likely to be affected adding to our view that this should be considered within the HRA.

Effects of the hydrazine plume on birds and their prey

3.561 Hydrazine will be discharged via the Construction Drainage Outfall (**CDO**) during the commissioning phase of construction as part of cold flush testing. This phase is likely to last several years and will result in a hydrazine plume with a mean sea surface area exceeding the chronic Predicted No-Effect Concentration PNEC of 30.5ha. The Shadow HRA Report[346] acknowledges that this has the potential to affect prey availability for SPA qualifying features (paragraph 8.8.279) and concludes that the peak instantaneous plume overlaps with 3.7% of the foraging range of little terns from the Minsmere-Walberswick SPA. No *adverse effects on integrity* is concluded on the basis of the small overlap and the short duration of the plume at 0.5 – 4.5 hours. For common tern of the Outer Thames Estuary SPA an overlap of 0.1% of the foraging range with the peak instantaneous plume above the acute PNEC is noted and again no *adverse effects on integrity* is concluded. A similar conclusion is reached for red-throated diver of the Outer Thames Estuary SPA, noting the small proportion of the SPA likely to be affected.

3.562 As the discharge of hydrazine during the commissioning phase of construction will be via the CDO, the hydrazine plume will form relatively close to the shoreline. Paragraph 22.8.367 acknowledges the potential for this plume to act as a barrier to migrating eels entering the Minsmere-Walberswick SPA via the Minsmere Sluice[347]. As eels form an important part of the diet of bittern (an SPA qualifying feature) this is of concern. Whilst we acknowledge that concentrations at the Sluice are expected to be well below levels at which toxic effects are expected, it is not clear whether the likelihood of impacts has been assessed under all potential tidal/weather scenarios.

3.563 We also note that paragraph 21.6.72 of ES Ch. 21 Marine Water Quality and Sediments[348] indicates that there is the potential for hydrazine to enter the Minsmere-Walberswick designated sites via Minsmere Sluice on ebb tides. It is explained that this potential is expected to be very low and that concentrations would be orders of magnitude below the PNEC.

3.564 However, given the sensitivity and high conservation value of areas that could be affected, further consideration should be given to possible scenarios under which even low levels of hydrazine could enter the system and any potential ecological effects. The Sluice contains a tide flap to the Leiston Main Drain which has a delay mechanism to allow tidal water to enter the Minsmere South Levels and maintain a brackish element to keep the Ramsar designated transition from brackish to freshwater within the ditches. The RSPB also open the Sluice to allow water northwards from the sea to maintain the brackish elements of the Minsmere Scrape. If hydrazine were to enter via the Sluice, the areas affected would therefore be Leiston Main Drain, the eastern section of Minsmere South Levels (including the brackish pool in the north east section regularly used by waterfowl, including gadwall and shoveler of the Minsmere-Walberswick SPA) and Minsmere Scrape, which is used by a variety of breeding, passage and wintering wildfowl,

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346 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

347 See Fig. 21.3 of ES Volume 2 Main Development Site Chapter 21 Marine Water Quality and Sediments Figures (APP-316)

348 ES Vol. 2 Main Development Site Ch. 21 Marine Water Quality and Sediments (AS-034)

waders, terns and gulls, again including birds of the Minsmere-Walberswick SPA, such as breeding avocet and little tern.

3.565 It appears that hydrazine is more persistent in soft, neutral/acidic water and it is our understanding that the spring water uprisings on the South Levels are soft and acidic meaning that the longevity of any toxic effects should be considered. We therefore consider that further detail of the potential effects of low-level hydrazine exposure to the relevant designated SPA and Ramsar features is required.

3.566 We note that paragraph 7.7.61 of the Shadow HRA Report[349] explains that hydrazine will be treated before release during the commissioning phase but that the treatment option had not at this stage been finalised. We request that this information is provided so that predictions regarding impacts can be confirmed.

3.567 During operation, the cooling system will be dosed with hydrazine on a daily basis for the lifetime of the development and a plume will form from the discharge point at the cooling water outfall. Two dosing strategies are considered, both of which result in the same annual load - 69ng/l for 2.32h a day and 34.5ng/l for 4.63h a day. The area of sea surface that exceeds the chronic PNEC is 158ha for the 69ng/l release and 157ha for the 34ng/l release, while sea surface exceedance for the acute PNEC is 17.4ha and 13.8ha for the 34.5ng/l and 69ng/l strategies respectively. For common tern of the Outer Thames Estuary SPA, the chronic PNEC overlaps with 3.2% of the mean foraging range of the birds nesting at Minsmere and the acute PNEC overlaps with 0.4% of this colony's mean foraging range. As this colony only forms part of the SPA population, no *adverse effects on integrity* is concluded.

3.568 For Sandwich terns of the Alde-Ore Estuary SPA, the chronic PNEC overlaps with 0.08% of the mean foraging range and the acute PNEC overlaps with less than 0.01% of the mean foraging range. For red-throated diver of the Outer Thames Estuary SPA 0.04% of the SPA is affected by the chronic PNEC concentrations and less than 0.01% for the acute PNEC values and no *adverse effects on integrity* is concluded.

3.569 Again, our concerns primarily relate to the combination of pressures on these species during the operational phase, including any potential impacts of hydrazine on prey within these species' foraging areas. However, we also query the potential for the hydrazine plume to have direct toxic effects on birds (during both commissioning and operation) and note that no evidence has been provided regarding this. Given that potential impact pathways exist, this impact should be considered within the assessment.

Direct and indirect impacts on birds from the sediment plumes

Introduction

3.570 Dredging will be required primarily during the construction phase for the construction of marine infrastructure (the Beach landing Facility (**BLF**), Construction Drainage Outfall (**CDO**), Fish Recovery and Return (**FRR**) and cooling water intakes and outfalls) and the maintenance of access to the BLF, but BLF maintenance dredges will also be required during operation. This will lead to increases in suspended sediment concentration (**SSC**) and in some cases the formation of a sediment plume. Our concerns relate to the potential for increased turbidity due to increased SSC to lead to avoidance behaviour by fish, particularly the prey species of SPA bird features and to lead to direct impacts on birds through avoidance behaviour by them or reduced hunting success (due to potential effects of increased turbidity on the ability of birds to forage successfully).

[349] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

Sources of increased SSC during construction and operation

3.571 Paragraph 22.6.66 of ES Ch. 22 Marine Ecology and Fisheries[350] explains the purpose, frequency and effects of capital and maintenance dredging required for the BLF:

> *"Navigational dredging for the BLF would include an initial capital dredge followed by maintenance dredging to maintain the navigable channel... Following the capital dredge, a plume with an instantaneous suspended sediment concentration (SSC) of >100mg/l above daily maximum background levels is expected to form inshore over an area of up to 108ha at the sea surface and 83ha as a depth averaged plume. A small area of up to 7ha would experience an instantaneous SSC plume of >1,000 mg/l above background levels…*
>
> *Assessments assume maintenance dredging of approximately 10% of the initial capital volume to occur at approximately monthly intervals during the campaign period when the BLF is in most frequent use... Maintenance dredging is predicted to result in up to 28ha of sea surface expected to experience >100mg/l, and 1ha expected to experience >1,000 mg/l above background SSC on each occasion…"*

3.572 Paragraph 7.3.10 of the Shadow HRA Report[351] explains that the plume resulting from the capital dredge is expected to form near to the shore and extend in a north-south direction and that it would be expected to disperse within four days. A relatively large area would be affected, as described above, although the plume is of short duration. Paragraph 7.7.56 of the Shadow HRA explains that:

> *"The average mean suspended particulate matter value at Sizewell from April to August is 31 mg/l (an average monthly maximum 80 mg/l) and during September to March is 73 mg/l (an average monthly maximum 180 mg/l)."*

3.573 The modelling of the impacts of dredging described above therefore indicates that significant uplifts from baseline SSC levels are expected, and whilst impacts from each individual dredge may be of short duration, the regularity of maintenance dredging during the construction (and operation) period and the additional dredging for construction of other marine infrastructure means that the potential for impacts on prey availability and accessibility for bird features of the Outer Thames Estuary and Minsmere-Walberswick SPAs must be considered.

3.574 Vol. 1 ES Addendum Ch. 2 Main Development Site[352] explains the proposals in the Change Application to enhance the permanent BLF and add a temporary BLF during the construction phase. Table 2.48 summarises the changes to dredging for the construction of the enhanced permanent BLF. This explains that the initial capital dredge volume and area are within the envelope of the original Application and therefore as no changes were required despite the permanent BLF being enhanced there was no further assessment needed. However, maintenance dredging is anticipated to increase in frequency but dredge volumes and thus suspended sediment concentrations would reduce. We query the level of increase in frequency and any potential for this to impact breeding birds in particular.

3.575 Table 2.52 explains that during operation:

> *"The advancement of the [enhanced permanent] BLF by 16m in comparison to the ES means more of the longshore bar will need to be removed. This is not necessary during the construction phase due to the installation of the grillage. The dredge volume estimate would*

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350 ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)
351 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
352 Vol. 1 ES Addendum Chapter 2 Main Development Site (AS-181)

be double the volume assessed in the Application…, therefore additional assessments are required."

3.576 Paragraph 2.17.105 goes on to explain that:

"A near doubling in dredge volume is anticipated based on the current bathymetry. This would result in instantaneous SSC plumes of 100mg/l above background over an area of several hundred hectares (compared to 83ha in the Application). In close proximity to the dredge area, SSC may exceed 1,000mg/l above ambient during dredging activities over tens of hectares (compared to 6ha in the Application)."

3.577 We note that paragraph 2.17.15 of Vol. 1 ES Addendum Ch. 2 Main Development Site explains that:

"the temporary BLF would cross the outer longshore sand bar. As such, there are no requirements for dredging and vessels could berth alongside with sufficient under keel clearance."

3.578 Dredging will also be required for the installation of the intakes and outfalls for the cooling water system. Paragraph 22.6.149 of ES Ch. 22 Marine Ecology and Fisheries states that for cooling water system dredging and dredge disposal:

"Plumes with instantaneous SSC of >100mg/l above background levels are expected to form over an area of up to 373ha (depth averaged, and 291ha at the sea surface). A smaller area of up to 14ha is expected to experience a depth averaged instantaneous SSC of >1,000mg/l above background levels (34ha at the sea surface)…"

3.579 The plume is expected to form parallel to the coast but not directly adjacent to it. Dredging for the installation of the cooling water infrastructure would be required six times and would not overlap (but could be consecutive), but SSC levels are expected to recover to background levels within two days each time.

3.580 Paragraph 22.6.163 of ES Ch. 22 Marine Ecology and Fisheries describes the dredging required for the FRR headworks and states that:

"Plumes with instantaneous SSC of >100mg/l above daily maximum background levels are expected to form over instantaneous areas of up to 89ha at the surface. A small area of 1ha is expected to experience an instantaneous SSC of >1,000mg/l above background at the sea surface".

3.581 A similar plume is expected to form during dredging for construction of the CDO. Again, the plumes would be expected to form parallel to the coast but would be closer to shore than those expected to arise from dredging for the cooling water system. Although impacts from dredging of the FRR headworks and CDO are of short duration (several days) and would only occur twice separated by around a year, as with the other dredges discussed, it will be necessary to consider whether these works could overlap with other marine impacts affecting prey species and therefore foraging birds.

3.582 Paragraph 22.6.176 of ES Ch. 22 Marine Ecology and Fisheries explains that the dredges required for BLF maintenance, installation of cooling water infrastructure and installation of the southern FRR headworks during the construction phase are likely to coincide. The plumes for the BLF maintenance dredge and the cooling water infrastructure are expected to remain as separate plumes, therefore affecting a greater area of the Greater Sizewell Bay at any one time. However, the plumes for BLF maintenance and the FRR dredge interact and increase the maximum

instantaneous area of sea surface exceeding 100mg/l to 111ha, although the duration of the plumes is reduced due to the temporal overlap. We also query whether there is any potential for further overlaps in dredging activities should construction timelines change due to poor weather or other delays. Impacts of any such scenarios should also be considered.

Impacts of increased SSC on birds

3.583 The potential impacts of the construction phase dredges on birds in terms of potential displacement of fish prey and the effects of increased turbidity on the ability of birds to forage successfully are not discussed in any detail. The Shadow HRA Report does discuss the overlap of the various dredge plumes with the foraging ranges of SPA species but should also consider the ecological consequences of this.

3.584 For breeding little tern of the Minsmere-Walberswick SPA, the extent of overlap of the foraging ranges of the Minsmere and Dingle little tern colonies is compared with the instantaneous SSC plume of over 100mg/l. Table 8.16 in the Shadow HRA Report[353] indicates that for the Minsmere little tern colony, the largest overlaps are 6% from dredging of the BLF navigational channel at spring tide, 3% for BLF dredging at neap tide, 2% for dredging for FRR1 at neap tide and 1% for dredging for FRR1 on a spring tide.

3.585 For the Dingle little tern colony, the largest overlaps are 4% for dredging for FRR1 on a spring tide, 2% for FRR1 on neap tides, 3% for dredging the BLF navigational channel on spring tides, 1% for BLF dredging on neap tides. The foraging range referred to in this table should be clarified – it is not stated whether this uses the mean or mean-maximum foraging range, and this would affect the areas quoted.

3.586 The potential for plumes to combine is also not illustrated in this table and, based on the points above, we would expect the total overlap for the combination of e.g. FRR and BLF dredging to be higher than indicated here.

3.587 The Shadow HRA Report concludes that no adverse effects on integrity on the Minsmere-Walberswick SPA will arise during construction on the basis that there is a relatively small overlap of the plumes and the little tern foraging range and the temporary/short term nature of the plumes and a similar conclusion is drawn for little tern of the Outer Thames Estuary SPA. However, we consider that the effects of combined and repeated SSC plumes have not been fully considered cumulatively or the potential ecological impacts on birds adequately explored.

3.588 For red-throated diver of the Outer Thames Estuary SPA, the Shadow HRA Report provides a limited assessment of potential impacts. Paragraphs 8.10.47-48 simply state that as the entirety of the SPA is available to foraging birds and the SSC plumes are relatively small and short-lived, no adverse effects on the SPA non-breeding red-throated diver population are predicted. Despite SSC plumes occurring within the SPA itself, no assessment has been undertaken of the potential effects on red-throated diver including birds being displaced and the ecological consequence of such displacement. Whilst we acknowledge that any displacement may be to elsewhere in the SPA, this still represents a loss in availability of part of the protected site.

3.589 In addition, even though each individual SCC plume may be relatively small and short lived, the effects of combined and repeated SSC plumes on red-throated divers have also not been fully considered nor any potential effects cumulatively with other potential effects such as loss of prey, thermal and chemical plumes (as discussed above).

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[353] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

3.590 As discussed above within the UK SPAs and SACs have tightly drawn boundaries, with no buffer zones and with everything included within that boundary being justified *ecologically*. Therefore, the potential for loss of availability of part of a protected site within the season for which it is designated, must be considered taking account of the protected sites' conservation objectives (as listed above) particularly

- The extent and distribution of the habitats of the qualifying features,…and,
- The distribution of the qualifying features within the site."

3.591 The potential impacts on birds arising from dredging for the enhanced permanent BLF during operation are given limited consideration in paragraph 2.17.131 of Vol. 1 ES Addendum Ch. 2 Main Development Site[354]:

> *"Fish may exhibit limited movements away from the areas of highest SSC, remaining in proximity to the plume and utilising the area once the plume dissipates. Localised and temporary displacement of sensitive taxa may occur. No significant changes in the availability of fish as prey items for designated features and as fisheries resources are predicted."*

3.592 While we acknowledge the sporadic nature of this impact, given the potential for this activity to coincide with key periods of the breeding bird season, we do not consider that it is possible to rule out all potential effects on the basis of the information presented. Exact figures for the area of the plumes are not presented and we request that these are provided and compared to the foraging ranges of SPA species to give an indication of potential impact.

Indirect impacts on birds from disturbance of prey species by underwater noise and vibration

Introduction

3.593 Paragraphs 8.8.311-312 of the Shadow HRA Report[355] explains that fish have varying levels of vulnerability to the effects of underwater noise and vibration, with potential effects including direct mortality, recoverable injury, Temporary Threshold Shifts (TTS) (temporary hearing impairment) or behavioural responses. Fish whose swim bladder is involved in their hearing are considered most vulnerable to negative effects of underwater noise and these include some of the main prey species for SPA bird species.

3.594 Underwater noise and vibration could therefore lead to reduced prey availability for birds if disturbance is significant enough to lead to mortality or displacement of their fish prey. Sources of noise and vibration include impact piling during the construction of the BLFs, dredging and drilling for construction of cooling water intakes and outfalls.

Impacts of noise disturbance on fish prey of little terns

3.595 A significant area of the foraging range of little terns from the Minsmere-Walberswick and Outer Thames Estuary SPAs (Minsmere colony) is expected to coincide with the area over which a fish 'behavioural response' (including displacement) is predicted. Table 8.17 in the Shadow HRA Report[356] shows that the overlaps are up to 46.5% of the colony's foraging range for the worst-case hammer energy scenario for piling of the BLF, with 29.1% of the range affected by the likely hammer energy scenario.

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354 Vol. 1 ES Addendum Chapter 2 Main Development Site (AS-181)
355 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
356 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

3.596 Dredging for BLF construction would affect 37.1% of the colony foraging range and if this coincides with dredging for the south cooling water intake this would affect 43.3% of their range. However, the Shadow HRA Addendum[357] describes the changes to predicted impacts as a result of construction of the enhanced permanent and temporary BLFs. Although the total number of piles has increased dramatically (from 12 to 148), this document makes a commitment to the avoidance of piling (and other BLF construction) activity during the bird breeding season, May to August. This would resolve concerns about noise disturbance from piling affecting foraging terns from the Minsmere-Walberswick and Outer Thames Estuary SPAs but requires all relevant details to be secured by way of a condition to ensure that this mitigation can be relied on and to justify conclusions.

Impacts of noise disturbance on fish prey of red-throated divers

3.597 Given that the Applicant has committed to avoid piling activity associated with construction of the BLFs during the bird breeding season, impacts on red-throated diver of the Outer Thames Estuary SPA during the non-breeding season must be considered. The Shadow HRA Report Addendum[358] states in paragraph 8.8.26 that:

> *"…Taking a worst-case in terms of underwater noise effects (i.e. as represented by the installation of the mooring dolphins and based on sprat, as a hearing specialist), the range of behavioural effect for the enhanced BLF (with mitigation) is predicted to cover an area of 1,206ha, representing 0.31% of the total SPA area. For the temporary BLF, the area is predicted to be 1,769ha, representing 0.45% of the total SPA area"*

3.598 It is concluded that no *adverse effects on integrity* will arise from indirect effects on the prey of red-throated diver due to the relatively short duration of the works, and because the area affected is limited to the north-west corner of the SPA where the Applicant asserts that diver concentrations are lower than the southern part of the SPA. As discussed above, we disagree with the assertion that this part of the SPA is of lower importance and we also note that the construction is expected to take place over two winters. We therefore consider that potential effects of displacement should be taken into account particularly when combined with other impacts on this species, for example, direct disturbance from vessel traffic.

Other potential noise/vibration effects on fish

3.599 We note that paragraph 2.7.2 of the Shadow HRA Report[359] indicates that tunnels would be created to around 3km offshore as part of the cooling water infrastructure. This tunnelling would take place around 30m below the seabed. Given the duration of the works (24 hour working for 15 months) we query whether any noise or vibration effects would be apparent within the water column which could affect fish distribution as this does not appear to be assessed within the Shadow HRA.

Direct disturbance of birds arising from vessel movements and other marine activity

Introduction

3.600 The revised proposals include the construction of permanent and temporary BLFs (described in 2b) ii) of Vol. 1 ES Addendum Ch. 2 Main Development Site[360]) to enable the delivery of Abnormal Indivisible Loads (AILs) and bulk aggregates during the construction period, with the permanent

[357] Shadow HRA Addendum (AS-173)
[358] Shadow HRA Addendum (AS-173)
[359] Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)
[360] Vol. 1 ES Addendum Chapter 2 Main Development Site (AS-181)

BLF also in occasional use during the operational period. In brief summary, we understand that the permanent BLF would be around 100m long, would operate in daylight only and be capable of receiving up to 100 beach landings per annual campaign (approximately 1 April to 31 October), which equates to 12-13 per month. Deliveries to the permanent BLF will be accompanied by two barges. The temporary BLF would be operational during the construction period only and located to the south of the permanent BLF and would be around 505m long, will operate both day and night and would be capable of receiving up to 400 deliveries per campaign, equating to 44 per month, or up to 600 per annum as this facility could additionally operate in the winter months. No additional vessels are required for deliveries to the temporary BLF.

3.601 Given the number of vessel movements expected, primarily during the construction period, our main concerns relate to direct disturbance of non-breeding red-throated divers. In addition, due to the presence of the two BLFs, and the length of the temporary BLF in particular, we are concerned about the potential for direct disturbance of these species from noise, lighting and visual disturbance from both the BLFs.

Potential disturbance impacts on non-breeding red-throated divers

3.602 As noted above, the proposals now include the potential to use the temporary BLF outside the campaign period of April to October (inclusive). This means that vessel disturbance may affect non-breeding red-throated divers of the Outer Thames Estuary SPA, likely to be present in Greater Sizewell Bay from around September to April. As discussed above, around 200 deliveries (400 movements) would be expected in each winter period (November to March inclusive), which gives an average of 2.7 movements per day.

Night-time vessel movements

3.603 The temporary BLF has the capacity to operate during the night as well as the day. Night-time use will require lighting of the temporary BLF and conveyor and we assume that this would be difficult to screen. This along with the noise associated with the operation of the enhanced BLF should be considered for potential impacts on loafing/roosting red-throated diver. Observations of red-throated diver from Thorpeness (Thurlow, 2009)[361] indicate that this species can have large roost movements in the early morning and suggest that the birds move from roosting to feeding areas in relation to tide. These indicate a need to understand the diurnal movements of red-throated diver given the need to assess impacts at night as well as daytime. Likewise, the indication that vessel movements will be influenced by tides and weather introduces a need to understand the impacts of tide and weather on the red-throated diver population to be able to adequately assess impacts.

Vessel movement corridor

3.604 Paragraph 8.8.16 of the Shadow HRA Addendum explains the assumptions made about vessels travelling through the SPA during the construction period. For the purposes of the assessment it has been assumed that vessels will originate from the Inner Thames meaning that they will pass through around 92km of the Outer Thames Estuary SPA to reach the BLFs at Sizewell. The vessel corridor will be approximately between two nautical miles and six nautical miles offshore, with the majority of vessels travelling through the SPA from the south and with those docking at the temporary BLF making their final approach from the north of Sizewell Bank.

3.605 Paragraphs 8.8.13-20 of the Shadow HRA Addendum go on to summarise the potential effects of the increase in vessel traffic associated with both BLFs. This assessment calculates the hours of vessel activity per km^2 of the SPA per month and concludes that the BLFs will only add 0.1 hours

[361] Thurlow, D. (2009) Wintering Red-throated Divers, Thorpeness, Suffolk 2000/01 – 2008/09. Suffolk Birds 2008.

of activity per km^2 per month. This appears to be averaged across the whole SPA area, while the figures quoted in paragraph 8.8.18 for comparison with existing levels of activity relate to specific vessel corridors, hence this assessment by comparison underestimates the impacts. It would be more helpful to calculate this uplift for the transit corridor within the SPA – at 92km long and 4km wide this would experience an uplift of nearly 1 hour per km^2 per month. This enables a more robust comparison with the statements about levels of movements in other specific corridors/areas within the SPA. Existing levels of use are stated to be between 1.5 and 10 hours per km^2 per month, with the busiest areas being within shipping lanes and around windfarms. The addition of a further 1 hour per km^2 per month would therefore represent an increase of 10% even in the busiest areas and an increase of 67% for the quieter areas. We therefore consider that the additional vessel activity due to the construction of Sizewell C is likely to significantly increase disturbance within the Outer Thames Estuary SPA.

3.606 We also note that we would expect to see greater certainty around the details of the proposals at this stage and request that more detail is provided regarding the vessel route. We would expect that such a route would attempt to minimise impacts on the SPA through use of existing shipping channels were possible, the grouping of vessel movements and route choice to minimise the time vessels spend in the SPA. These details should be provided in a vessel management plan.

Sensitivity of red-throated divers to vessel movements

3.607 In assessing the impacts of the additional activity described above the Shadow HRA Addendum states that any disturbance is likely to be short lived and temporary with birds likely to return after the vessel has passed. We disagree with those assumptions as red-throated divers are known to be particularly sensitive to disturbance, including that from vessel movements. Mendel *et al.* (2019)[362] assessed the effects of offshore windfarms and shipping on the distribution and density of diver species (predominantly red-throated diver) in the German North Sea. They found significant reductions in abundance of divers around windfarms and around shipping areas and noted that one in three divers are likely to leave an area as a ship approaches, resulting in a significant reduction in diver densities within 5km of vessel activity. It is also notable that no habituation to the presence of windfarms was observed after 3 years. Schwemmer *et al.* (2011)[363] also reported high avoidance of shipping lanes by red-throated divers, noting that habituation to vessel disturbance is unlikely to occur and categorised the flush distance of red-throated divers in response to vessels as "very high". Irwin *et al.* (2019)[364] also noted strong avoidance of shipping lanes by red-throated diver in their aerial surveys of the Outer Thames Estuary SPA.

Further assessment needed

3.608 Paragraph 8.8.20 of the Shadow HRA Addendum suggests that the increase in vessel traffic arising from the enhanced permanent BLF and the addition of the temporary BLF will not give rise to *adverse effect on integrity* of the Outer Thames Estuary SPA. Little evidence is presented to justify this conclusion. We consider that assessment of the effects of displacement arising from vessel movements on red-throated divers from the Outer Thames SPA should be carried out following the standard methodologies used for offshore windfarm and cable route assessments and more

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362 Mendel, B., Schwemmer, P., Peschko, V., Müller, S., Schwemmer, H., Mercker, M. & Garthe, S. (2019) Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of Loons (Gavia spp.), Journal of Environmental Management. 231: 429-438.

363 Schwemmer, P., Mendel, B., Sonntag, N., Dierschke, V., & Garthe, S. (2011) Effects of ship traffic on seabirds in offshore waters: implications for marine conservation and spatial planning. Ecological Applications 21:1851–1860.

364 Irwin, C., Scott, M., S., Humphries, G. & Webb, A. (2019) HiDef report to Natural England - Digital video aerial surveys of red-throated diver in the Outer Thames Estuary Special Protection Area 2018. Natural England Commissioned Reports, Number 260.

detailed consideration of potential mortality (through increased mortality elsewhere in the SPA) is needed. We suggest the following approach to the assessment:

1. Application of 5km buffer around the BLFs and vessel routes (based on recent evidence, including Mendel *et al.*, 2019).

2. Assessment of scale of possible displacement. Present a matrix showing a range of displacement percentages (0% to 100%) and resulting percent mortality (0% to 100%), then review relevant offshore wind farm cases (e.g. Kentish Flats – Percival 2010[365], and Horns Rev – Petersen *et al.*, 2006[366], London Array – APEM, 2021[367]) to indicate what might be the most realistic percentage displacement. The mortality consequences are unknown and so it will be necessary to consider the full range of possibilities. Having determined the sensitivity of divers to different levels of displacement and additional mortality, the next step will be to determine the need for population modelling.

3. Potential implications for the Outer Thames Estuary SPA population may require population modelling using a density independent model[368]. There is no evidence for density dependence operating on wintering red-throated divers and no suitable measures of density dependence are available to apply, so the outcome of an assessment using a density dependent model could be meaningless. Therefore, a density independent approach is arguably more appropriate, and would be precautionary.

3.609 Points 2 and 3 relate to mortality impacts arising through displacement. However, the maintenance of diver abundance is not the sole Conservation Objective of concern for this species within the Outer Thames Estuary SPA with regard to the HRA. The Conservation Objectives to maintain diver distribution and to maintain the extent of supporting habitat are also likely to be compromised as divers will redistribute within, and outside, the SPA due to the disturbance from shipping. This redistribution compromises the ability of the supporting habitat within the SPA to continue to support the same numbers and distribution of birds as before (Natural England, 2020[369]). As such, these should also be included in the consideration of whether *adverse effects on integrity* can be ruled out.

Population data for assessment

3.610 With respect to the local population numbers of red-throated divers, as these birds move around based on food supply, during some periods in some years a substantial number of divers may be present in the area and potentially displaced. We therefore advise that a reasonable worst-case scenario should be based on such local maxima. However, it should be noted that the data

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365 Percival, S. (2010) Kentish Flats Offshore Windfarm: Diver Surveys 2009-10. Report on behalf of Vattenfall Wind Power.

366 Petersen, I., Christensen, T., Kahlert, J., Desholm, M. & Fox, A. (2006) Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark. Natural Environment Research Institute.

367 APEM Ltd (2021) Final Ornithological Monitoring Report for London Array Offshore Wind Farm – 2021. (Document available by searching the MMO public register (here) for case ref 32945/050608/39 and viewing the return ref 1.4 and doc P00002714 LAL Year 3 Post Construction Final Reporting_v4.0_Clean.pdf)

368 Density dependence refers to processes which act on a population as it grows to regulate its growth rate. Density dependence may include factors such as competition between individual birds for resources and disease which become more prevalent with a larger population. However, whilst density dependence is known in many bird populations, species and site-specific evidence about the strength of such factors is often lacking, meaning that for the purposes of population modelling the most precautionary approach is usually to use a density independent model (unless evidence about the type and strength of density dependent factors acting on a particular population is available).

369 Natural England (2020) Appendix A12 to the Natural England's Deadline 4 Submission NE advice on Red-Throated Divers in the Outer Thames Estuary Special Protected Area related to Deadline 3 submissions for East Anglia ONE North Offshore Windfarm.

provided in Annexes to ES Ch. 14 Appendix 14A7 Part 2[370] are based on vantage point surveys based along the coastline.

3.611 We do not consider that vantage point surveys alone provide a robust methodology as data are required up to 3km (preferably 4km) to seaward of the shipping channel. Viewpoints on a low coast are very unlikely to reliably accomplish this and may be less likely to record birds on the water than those in flight. A correction factor will also be required to account for the number of birds likely to have been diving at any time. Therefore, while the survey data do indicate the importance of Greater Sizewell Bay for red-throated divers, the resulting population figures may be underestimates.

3.612 These surveys also only cover the Greater Sizewell Bay area whereas the whole vessel route through the SPA requires consideration. Aerial surveys in 2013 have shown red-throated diver densities in the SPA have increased and indicate that previous surveys may have underestimated numbers (Irwin *et al.,* 2019[371]). The species is known to be sensitive to flushing by boats (Schwemmer *et al.*, 2011[372], Jarrett *et al.*, 2018[373]) and recent surveys have been conducted using aircraft, originally with observers on board to count birds in strip or line transects and most recently recording high resolution digital imagery using either still photography or video footage. We recommend that these data should be used to inform the assessment of the relative usage of this area by divers and the propensity for this species to be disturbed by boats needs to be taken into account in the impact assessment.

Conclusion

3.613 In conclusion, we disagree with the approach to the assessment of impacts of vessel movements on red-throated divers of the Outer Thames Estuary SPA and the assumptions made about responses of divers to disturbance. We recommend that additional assessment of displacement effects is carried out. We consider that vessel disturbance impacts from Sizewell C could compromise the SPA objective to maintain the distribution of the qualifying features within the site and therefore do not support the conclusion of no *adverse effects on integrity* in the Shadow HRA Addendum.

Combined marine impacts

3.614 We are concerned that the combined effects of disturbance resulting from dredging, piling and vessel movements, the various discharges (including those of thermal discharges, bromoform, hydrazine and organic matter (dead fish and other organisms)) and increased suspended sediment concentrations and the resulting total displacement of marine birds (particularly red-throated diver of the Outer Thames Estuary SPA) have not been fully considered. We acknowledge that these issues are covered in the Shadow HRA Report Addendum Appendix 1A[374] but we consider that this does not advance understanding much beyond the assessment of the individual impacts. For example, for combined marine impacts on red-throated diver, little and

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370 ES Vol. 2 Main Development Site Ch. 14 Terrestrial Ecology and Ornithology Appendix 14A7 Ornithology Part 2 of 2 (APP-238)

371 Irwin, C., Scott, M., S., Humphries, G. & Webb, A. (2019) HiDef report to Natural England - Digital video aerial surveys of red-throated diver in the Outer Thames Estuary Special Protection Area 2018. Natural England Commissioned Reports, Number 260.

372 Schwemmer, P., Mendel, B., Sonntag, N., Dierschke, V., & Garthe, S. (2011) Effects of ship traffic on seabirds in offshore waters: implications for marine conservation and spatial planning. Ecological Applications 21:1851–1860.

373 Jarrett, D., Cook, A. S. C. P., Woodward, I., Ross, K., Horswill, C., Adam, D., and Humphreys, E. M. (2018). Short-Term Behavioural Responses of Wintering Waterbirds to Marine Activity: Quantifying the Sensitivity of Waterbird Species during the Non-Breeding Season to Marine Activities in Orkney and the Western Isles. Scottish Marine and Freshwater Science Vol 7 No 9, 88pp.

374 Shadow HRA Report Addendum Appendix 1A Further Analysis of Inter-Pathway Effects (AS-174)

common terns of the Outer Thames Estuary SPA during construction and operation, paragraph 3.6.6 of the Shadow HRA Addendum Appendix 1A states that:

> *"… marine water quality effects during construction and decommissioning and disturbance effects on species populations during operation are essentially inconsequential. Therefore, inter-pathway effects could only occur via the pathways for the marine water quality effects and interaction with project infrastructure during the operational phase. These pathways both have the potential to affect the foraging conditions and / or food availability for the qualifying features of the Outer Thames SPA. However, the effects from both pathways on each of the qualifying features are predicted to be small and consequently no adverse effect on site integrity is predicted when the respective effects are considered together."*

3.615 As we consider that the individual effects have been underestimated, particularly of disturbance by vessel traffic and impacts on prey species, we do not agree with this conclusion. No discussion of the potential for interactions is provided, for example through consideration of effect timelines and seasonality and no discussion of the consequences for fitness and survival of these populations is included. Similar conclusions are reached for common and little terns of the Minsmere-Walberswick SPA.

3.616 Known synergistic issues have also not been covered in sufficient detail. Paragraph 22.6.391 of ES Ch. 22 Marine Ecology and Fisheries[375] acknowledges that toxicity of chlorinated discharges to many marine organisms increases with temperature, meaning the potential exists for synergistic effects of the thermo-chemical plumes on fish. This increase in toxicity with temperature also applies to the hydrazine discharge. It is concluded that whilst toxicity may be enhanced, this would only affect small areas and would be insignificant in the context of natural variability. The effects are not expected to extend beyond the existing TRO EQS contour (as the largest of the chemical plumes likely to have synergistic effects with thermal uplift), although as this covers 338ha as a 95th percentile, the area affected could be relatively large. Potential ecological effects (e.g. on birds as predators) are not explained.

Monitoring

3.617 Given the potential for impacts on bird distributions arising from the combination of marine impacts, monitoring of these populations during the construction period will be required. At present, we note that no such monitoring is proposed and that monitoring of direct impacts (e.g. of discharges on water quality, of impingement and entrainment numbers) is covered through a number of separate routes. We are therefore concerned that at present, there is no mechanism to detect ecological impacts of the combined marine activities and discharges, particularly on bird features of the Minsmere-Walberswick and Outer Thames Estuary SPAs. We recommend that an appropriate monitoring package is developed to verify impact predictions and identify the potential need for mitigation.

Marine impacts in-combination with other projects

3.618 Potential exists for in-combination effects on red-throated divers of the Outer Thames Estuary SPA from the Sizewell C project and several offshore windfarms, particularly during the construction phase. For example, the cable routes for the East Anglia One North and East Anglia Two projects pass through the Outer Thames Estuary SPA, vessels associated with the projects will transit through the SPA and part of the 4km buffer (for assumed bird displacement impacts) for East Anglia One North is also within the SPA. Given that the construction of Sizewell C could coincide with construction and operation of these projects and others, in-combination

[375] ES Vol. 2 Main Development Site Ch. 22 Marine Ecology and Fisheries (AS-035)

disturbance and displacement of marine birds is likely. Table 8.30 in the Shadow HRA Report[376] concludes that no AEOI of the Outer Thames Estuary SPA will arise as impacts of these two windfarms are each predicted to be small and vessels will stay within existing shipping routes. However, we again consider there is a need to look at sum total of disturbance impacts affecting this SPA in the style of an offshore windfarm assessment of displacement. This would consider disturbance displacement from Sizewell C and all relevant windfarms together rather than the approach taken by this assessment, which only considers Sizewell C plus each windfarm individually and will significantly underestimate in-combination impacts.

3.619 We also note that the assessment of SzC in-combination with other projects in Section 4.15 Marine Ecology and Water Quality of ES Vol. 10 Ch. 4 Assessment of Cumulative Effects with Other Plans, Projects and Programmes[377] does not consider impacts on marine birds. This is a key omission, particularly with regard to in-combination impacts of vessel traffic on red-throated divers.

Conclusions regarding marine impacts

3.620 In conclusion, based on the above concerns, we do not agree that AEOI can be ruled out for the following sites and species for impacts from the project alone:

- Breeding little terns of the Minsmere-Walberswick SPA
- Non-breeding red-throated divers of the Outer Thames Estuary SPA
- Breeding Sandwich terns of the Alde-Ore Estuary SPA

3.621 In relation to this project in combination with other projects, we do not agree that AEOI can be ruled out for the following site and species:

- Non-breeding red-throated divers of the Outer Thames Estuary SPA

Bats

Introduction

3.622 Our response focusses on two bat species, primarily barbastelle and secondarily, Natterer's bat. Barbastelle are found only in southern England and Wales. Very few breeding sites are known (Sizewell estate being one) and the species is protected in the UK under the Wildlife and Countryside Act, 1981. It is also a Priority Species under the UK Post-2010 Biodiversity Framework. Within the European Union, it is a Protected Species under Annex IV of the European Habitats Directive. Internationally the rarity of barbastelle is recognised, being listed as Near Threatened on the global IUCN Red List of Threatened Species. Natterer's bats are protected in the UK under the Wildlife and Countryside Act, 1981. The species is widespread, but scarce in the UK.

3.623 Following Wray *et al* (2010)[378], the Sizewell barbastelle population on the Sizewell estate is recognised as being of national importance and the population of Natterer's bat, as of county importance.

3.624 Much of the content of this response draws on the findings and conclusions of two reports by BSG consultants, commissioned by East Suffolk Council and Suffolk Wildlife Trust. These reports have already been submitted into the examination as part of the Local Impact Report: Joint Local

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376 Shadow HRA Report Volume 1: Screening and Appropriate Assessment Part 1 of 5 (APP-145)

377 ES Vol. 10 Project-wide, Cumulative and Transboundary Effects Ch. 4 Assessment of Cumulative Effects with Other Plans, Projects and Programmes (APP-578)

378 Wray, S., Wells, D., Long, E., & Mitchell-Jones, T. (December 2010) Valuing Bats in Ecological Impact Assessment. Chartered Institute of Ecology and Environmental Management In-Practice 70: 23-25

Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092]. The majority of the concerns raised below are directly related to findings and concerns within these reports. Where we have made additional points of concern, we have clearly indicated this.

Summary of Concerns

3.625 The following section is a summary of our main concerns over potential impacts on bats, the full details of which can be found in the main bat section following the summary.

Hinkley C

3.626 There is repeat reference of Hinkley C as an analogous example. The footprint of Hinkley C is dominated by an intensively managed mixed farmed landscape, likely supporting low levels of bat activity, whereas the Sizewell estate includes a wetland SSSI, mixed woodland and a wider woodland/wetland complex in close proximity to RSPB Minsmere, supporting a nationally important population of barbastelle and Natterer's population of county importance. Therefore, Hinkley C does not have a comparable bat population from which to base conclusions on potential impacts or effectiveness of mitigation;

3.627 There is very little evidence of roosting within the footprint at Hinkley and therefore it would be expected to have far lower levels of activity due to the poor quality foraging habitats;

3.628 The combination of survey and monitoring at Hinkley is only robust enough to say there continued to be some use of a retained feature across the construction site (a green lane) by barbastelle during construction. The key factors such as number of bats, timing and frequency of movement and the nature of this use (such as foraging and commuting) are not commented on.

Data Adequacy and Analysis

3.629 The removal of static recorders from the footprint of the development site means there is a loss of baseline data that makes true impact impossible to determine;

3.630 There is a distinct lack of bat data within the area of Sizewell Marshes SSSI that will be lost to the development;

3.631 There is a lack of data from Goose Hill. This has led to a general risk that the importance of Goose Hill has been significantly underestimated in terms of roost provision as well as usage by pregnant females and juvenile barbastelle;

3.632 There is a lack of quantitative data, such as thermal imaging in the dataset, making it impossible to determine actual numbers. This in turns affects the quality of the data analysis. For example, it will be a lot more difficult to monitor actual changes in use that could be linked to population effects;

3.633 There is limited temporal coverage across the dataset, especially during June for the 2020 dataset. Only half of the detector locations (28) were subject to sampling. This is the month in which barbastelle activity on the EDF Estate has been highest and is clearly key in the context of the locally-breeding population;

3.634 There is not enough emphasis on rarer species within the analysis of the data. The use of a percentage metric means common species mask the actual (rather than relative) importance of a specific location for rare species;

3.635 The way in which the data have been analysed is leading towards a generic monitoring protocol that is unlikely to pick up population changes in rarer species such as barbastelle;

3.636 There are no conclusions as to what the predicted residual effects may be for barbastelle. Having concluded significant impact on barbastelle due to habitat fragmentation, there appears to be no attempt to explain what that will actually mean to the population or, how conclusions can be tested through an appropriately detailed monitoring protocol;

3.637 There can be no confidence that the habitat creation will effectively compensate habitat loss (supporting the conclusion in the ES of a minor adverse (not significant), as the data are not used to underpin a detailed evaluation of this;

3.638 Overall, the generic approach i.e. to look at the community broadly as opposed to the key species is of considerable concern. The likely consequence of this is that there is limited understanding of how Natterer's bat and barbastelle populations will respond to the construction. Any age / sex or life-stage related effects are likely to be missed.

Noise

3.639 There are very little data on the impacts of noise on bats and therefore a more precautionary in the approach to assessment should have been adopted;

3.640 Acceptable levels of noise in relation to features (such as roosts and commuting routes), have not been defined. It is unclear why the developer needs to wait to do this and what is practical and achievable;

3.641 There is no confidence that where noise exceeds thresholds (if there were any), that impacts on bats can be effectively mitigated as there are no clearly defined mitigation measures are in place;

3.642 The likely effectiveness of key mitigation approaches such as the hording or bunding at Ash Wood remains unclear and the practicalities of how monitoring will establish its effectiveness are not explored;

3.643 The threshold for licencing is not identified and no indication of how it might be derived is provided;

3.644 There is no practical framework from which an Ecological clerk of Works (ECoW) team can derive a working system that will identify exceedances and allow the implementation of controls;

3.645 In addition to concerns identified in the consultant's report we consider:

- Clarification is required to determine that noise is being properly assessed, with a focus on frequency that is ecologically relevant to bats, rather than the use of human noise modelling;

- There appear to be discrepancies between ES chapter AS-033 and the updated bat impact assessment AS-208 despite both referring to noise modelling data and results presented in the Bat Mitigation Strategy (APP-252 and APP-253). It is unclear which is correct and hence it is impossible to determine noise impacts in some key areas;

- Concerns over spatial coverage of monitoring with limited coverage in many key areas including, but not restricted to; Ash Wood, the southern end of Bridleway 19, Fiscal Policy, along Kenton Hills and areas bordering the SSSI;

Lighting

3.646 Overall, it is difficult to take an independent view on the approach to and conclusions of the assessment based on the limited information provided. The combination of the acknowledged lack of certainty in the assessment and the indication that it may not be practically possible to implement further mitigation adaptively then leads to the conclusion that ultimately effects of lighting on bats will be a secondary concern during construction;

3.647 We therefore consider the assessment should be more precautionary;

3.648 There is a failure to define acceptable thresholds of light at key features;

3.649 There is a lack of clarity over lighting of the Sizewell Marshes and light spill from the bridge to cross the SSSI;

3.650 The use of the phrase 'reasonably practicable' with regard to lighting. This and other statements suggest that health and safety will determine lighting levels during construction, and implementing adaptive mitigation for impacts on bats may be impossible in practice;

3.651 In reference to the above point, there is no indication of a practical framework from which an ECoW team can derive a working system that will identify exceedances and allow the implementation of controls.

3.652 In addition to concerns identified in the consultant's report we consider:

- The rationale for choosing the locations for modelling light levels are unclear. There are concerns over the spatial coverage of light level in the baseline survey especially around Ash Wood, at Upper Abbey Farm, Fiscal Policy, along Bridleway 19 especially the southern end near the plaza and security areas, alongside Kenton Hills especially opposite green rail offload area and areas bordering the SSSI;

Evaluation of the value of Goose Hill

3.653 We are concerned over the conclusion that there is 'minimal roosting resource for bats' within Goose Hill when the data show the total loss of 111 trees of moderate and high roost potential. We believe this conclusion may have led to a significant underestimation of the effect the loss of Goose Hill will have, especially on barbastelle. Climbed assessment is required to confirm the overall roost potential;

3.654 The foraging and commuting value of Goose Hill has not been properly mitigated.

3.655 The importance of Goose Hill for breeding females and juvenile barbastelle has not been recognised and the effect displacement will have on their foraging, given their ability to fly longer distances~~,~~ is reduced. This is the sort of issue that will drive potential changes in populations / their potential relocation.

Mitigation

3.656 We are concerned that the development has reached an advanced stage in the planning application process, with almost 15 years of data to draw on, but there are no detailed mitigation proposals or a structured monitoring plan in place. It is difficult for stakeholders to be asked to comment on these very detailed assessments without the details of the mitigation that allow the conclusions in the ES to be reached;

3.657 There is a need to overlay both noise and light contours over the mitigation areas and core bat areas that apparently are un-impacted (Ash Wood, Bridleway 19, Fiscal Policy, Kenton Hills, the SSSI triangle, the SSSI west of platform, the proposed 'dark' corridors, areas around the Lower Abbey Farm bat house and around Upper Abbey Farm);

3.658 Impacts from task specific lighting are uncertain due to statements such as, 'if safe to do so' and 'where reasonably practicable'. More certainty is required and if this cannot be given, then the proposal has to assume impacts based on the precautionary principle and hence a need for mitigation;

3.659 Concerns over limited amount of buffering of light spill in key roosting and foraging areas;

3.660 Uncertainty and lack of detail with regard the type of lighting that will be used and how this will inform mitigation;

3.661 There appears to be a limited number of bat boxes. Given the low level of effort put into searching for roosts, especially within the SSSI, it seems highly likely that the number of bat boxes needed to sustain barbastelle, has been significantly underestimated. Additionally, given the most important bat roost found on the entire estate is a dead conifer, some thought should be given to veteranisation of a variety of trees species in suitable locations.

3.662 There appears to be no commitment to monitor 'dark' corridors or the SSSI culvert during the development to determine use and hence effectiveness of the mitigation proposed.

3.663 A monitoring protocol needs to consider the ecology of the key bat populations and how they behave at different ages and stages in the breeding cycle. It needs to consider what are the actual impacts on barbastelle and Natterer's bat and how they reflect those predicted in the ES.

3.664 In addition to concerns identified in the consultant's report we consider:

- We have concerns over the functionality of the new bat house at Lower Abbey Farm as a result of it being cut off from commuting routes due to light and noise;

In-combination (Synergistic) Effects

3.665 Little has been done to understand the combined impacts of light, noise and fragmentation together;

- There is an underestimation of the importance of the landscape affected by the footprint of the Sizewell Link Road but additionally how this area may increase in importance during the development as bats are displaced from the main development site. The assessment states very low levels of use of hedgerows along the road route by barbastelle. This is not the case if the data set is reviewed in the context of the data set for the main development site. Some of the highest rates of encounter of barbastelle from static detectors have been noted along the potential route.

Data Adequacy

Spatial Coverage

3.666 There is concern about the removal of many of the static bat monitoring points during the 2020 data collection and especially MS10, MS8, MS11, MS16 and MS32. The statements explain the reasons for removal are because the sites are, *'lost under construction footprint therefore this is not an appropriate location to assess against future construction impacts'* (static monitoring report, Table 4, AS-037). These monitoring sites include the hedge line north from Kenton Hills,

the junction of Stonewall Belt and Hilltop Covert and the eastern edge of Goose Hill close to the SSSI. All of these locations appear to be areas of high barbastelle use in the past, so it would be expected these areas need to be included to determine a pre-construction baseline. Consequently, the conclusion for removal of these sites is difficult to explain or justify if the aim is to understand impacts of the development on bats. It is noteworthy that previous surveys from 2013 and 2014 (static monitoring report, Table 1, AS-037) have all recorded commuting or foraging bats at these locations and hence further data would have helped establish a robust baseline from which to determine overall use of the estate around and within the development. Without these data it will be difficult to determine how overall bat use has changed once construction starts.

3.667 The 2020 Backtracking Report (AS-037) section 2.3.6 states, *'Tree roost potential assessments were conducted on trees likely to be impacted by the works to inform the potential for impacts upon bat roosting within trees'.* However, the baseline dataset does not include any bat monitoring within the SSSI 'triangle' itself i.e. the area of the SSSI that will be lost under the SSSI crossing. Consequently, this statement is factually incorrect. This omission is likely to be significant especially in light of the comment in 2.3.7, *'Barbastelle commuting has been noted with individuals commuting north to south (in the direction of Sizewell Marshes SSSI)'.* This creates significant uncertainty that the dataset is a true reflection of the baseline in terms of commuting and foraging, risking inadequate mitigation measures related to noise and lighting spillage into the SSSI. Furthermore, given the significant tree cover in this area, it is possible there will be multiple loss of roosts that are not recorded and therefore not mitigated.

3.668 There is very limited monitoring to the west of the platform within the SSSI and to date, there appears to be no roost assessments in this area. MS29, the one monitoring station within the locality appears to be very important for rare bats and so the paucity of data from this area of SSSI, which will be lost under the footprint of the development, is a significant omission.

3.669 The 2020 data (AS-037) show clearly that barbastelle use Goose Hill and Fiscal Policy but there are no data or analysis that help with frequency of use to determine numbers, or direction of flights. Overall, there is very limited monitoring within Goose Hill and this appears to be a significant omission, given the evidence that is available, which suggests the area may be very important for barbastelle (please see the Section 1.7 for further detail).

3.670 Plot 4 in the static monitoring report (AS-037), which shows the location of the static monitoring sites, only has 25 of the 29 locations marked. A notable omission is MS06 at NE corner of Ash Wood. This location had a barbastelle maternity colony from previous work and so is an important site to have omitted.

3.671 There are no static detectors located in open ground which appears to be an omission given barbastelle are well known for commuting over open ground. This may result in an underestimation of the overall importance of the development site for barbastelle.

Temporal Coverage

3.672 Temporally there is a substantial recent data gap for barbastelle. The 2020 static survey data covered three months but with limited data from June (Table 5, AS-037), despite Table 2 (AS-037) clearly showing in previous years static monitoring work that the average barbastelle activity was the highest in this month. In the report it is stated that static detectors could not be deployed before 18 June but there is no explanation of why or any acknowledgement that this gap will invariably create more uncertainty in conclusions.

3.673 Back tracking surveys at one survey per month (Table 4, AS-037) in densely wooded habitat which is inherently difficult to survey, seems relatively low effort especially during the maternity period. Again, this will create further uncertainty in any conclusions.

3.674 The temporal coverage does not comply with good practice guidelines for survey effort (Collins, 2016)[379]. Table 8.3 recommends static detector surveys between April/May and September/October. Section 8.4.7 recommends back tracking surveys should be continued until roosts are located unless alternative methods are considered more appropriate.

Data Collection Methodology

3.675 Overall, there is a lack of complementary observational data, such as from thermal imaging cameras which would help to quantify bat use in any given area. Currently the nature of the data means there is no way of telling the numbers of different individual bats of a specific species using an area. In static surveys there are no links between the number of bat calls and actual bat numbers. For example, a high number of calls could be from a single bat, conversely a low number of calls could be from a number of bats moving through an area. Despite the limitation in the data, there is very little recognition of this fact. Overall, this calls into question the use of the data and how useful it will be to inform a baseline for a future monitoring protocol.

3.676 The Environmental Statement notes that too few individual Natterer's bats were radio tracked to enable home ranges to be identified. This is a notable omission but despite this, little attempt has been made to draw conclusions from the available evidence of radio tracked data or comment on differences between age classes.

Data Analysis

3.677 Throughout the assessment there is not enough detailed analysis on the rarer species. Data analysis has focussed on cumulative population as opposed to single species activity. The design of the static work means that it will only be possible to determine overall volume of use with little idea on actual numbers of bats to enable year-on-year comparisons. This limitation in the dataset is further weakened by the fact that the nationally important population of barbastelle and county level important population of Natterer's bat are reflected as a percentage of the bat population as a whole. This is a common error in the treatment of ecological data. The use of a percentage as a metric risks rare species being masked by more common bats. For example, where the locations of soprano pipistrelle are under 90%, this is used to say where the species rich sites are. However, this might merely be a reflection of the fact that soprano pipistrelle don't like that particular location; calls from rare species in total could still be lower. Conversely a really species rich site could have many soprano pipistrelle, rendering rare bats low as an overall percentage of the population, despite having high numbers and being more important for barbastelle and Natterer's bat. The use of percentage as a metric only weakens what is already a poor quality dataset. The lack of quantitative data means the failure to capture static detector data at 14 locations in June appears all the more significant.

3.678 In conclusion, a survey protocol designed to provide a high resolution baseline has not been achieved because: a) the specification of the data are not of high enough resolution to allow a reliable identification of changes of behaviour that could lead to a determination of population level effect; b) There has been inadequate consideration of the nationally important barbastelle population and county level important Natterer's bat population; c) too much weight and consideration has been given to more common species.

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[379] Collins, J. (ed.) (2016) Bat Surveys for Professional Ecologists: Good Practice Guidelines (3rd edn). The Bat Conservation Trust, London.

3.679 The overall result of this is a lack of barbastelle monitoring and focus with incoherent conclusions in terms of the importance of the Sizewell estate to this species.

Future Monitoring

3.680 Overall, no conclusion is drawn on what the predicted significant residual effect of habitat fragmentation on barbastelle will mean for the population. Understanding how the population is likely to respond is key to understanding the likely outcome for the species and hence in formulating a monitoring strategy. Given the fact that no conclusions have been drawn on what the predicted significant residual effects might be is of significant concern, compounded by the fact that no monitoring strategy has been provided.

3.681 Overall, the monitoring detail is limited[380] and if the development is consented based on the current information, then there will be a great deal of pressure from construction timelines to come up with a monitoring protocol. In our view, there needs to be a condition in place on the development that a detailed Bat Monitoring Strategy is produced prior to any construction taking place, with clear objectives, as a headline of maintaining a breeding population of barbastelle. This should include protocols for monitoring, method statements, identification of triggers for light and noise impacts (for further mitigation) and evidence based examples of possible future mitigation and must be secured through the DCO. Funding for possible future mitigation must be secured through the DCO.

3.682 Future bat monitoring at Sizewell should focus on understanding the drivers for development-related population change in key species (barbastelle and Natterer's bat) and should be of high enough resolution to achieve this. It is concerning that current bat monitoring locations do not include roosting barbastelle and Natterer's bat at Goose Hill, Ash Wood or Fiscal Policy. In our view further survey work needs to be done in these areas in addition to furthering baseline work within the SSSI (see below).

3.683 Despite predicted residual significant impact on barbastelle as a result of fragmentation, there is no explicit commitment to monitor the SSSI crossing for usage, the effectiveness of the retained dark corridors along the Upper Abbey Farm bridleway or the central corridor running north from Kenton Hills. Furthermore, there is no commitment to monitor the usage of the newly formed habitat of arable reversion once construction is complete, which, it is assumed, will be used by bats for forage.

3.684 If the nature, scale and duration of a development are such that noise could cause disturbance to important bat populations, then unweighted, high-frequency noise measurements of existing ambient levels based on frequencies audible to bats (~8 kHz upwards) should be collected (Reason and Bentley, 2020)[381] at these locations. This should continue during construction to monitor the bats' response to noise.

3.685 As outlined above, it is concerning that the static survey work of 2020 was not complemented by other survey work to give an indication of bat numbers. Therefore, it is unclear how monitoring can be designed to detect changes in bat commuting and foraging resulting from the development and hence is likely to be unable to detect population level effects. Comparing generic activity levels is less relevant especially as much of the work has focussed on commoner species.

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[380] Terrestrial Ecology Monitoring and Mitigation Plan (REP1-016)

[381] Reason, P., & Bentley, C. (June 2020) Noise Impacts on Bats – A Sound Assessment? Chartered Institute of Ecology and Environmental Management In Practice 108: 15-18

3.686 Volume 2 Main Development Site Chapter 11 Noise and Vibration (Doc Ref 6.3) 11.1.1 [App-202]) states routine monitoring will be carried out during construction in accordance with the Code of Construction Practice (CoCP) AS-273. The CoCP outlines noise monitoring for human receptors, not for bats and therefore it appears that noise monitoring will not be at ecologically relevant frequencies (8kHz and above), or at heights used for foraging. Hence we have significant concerns over adequacy of monitoring and mitigation. The noise levels in key areas of the site for bats must be monitored at 8kHz and at 10m above ground level throughout the construction phase. This is discussed in more detail in following sections.

Noise

Use of Human Noise Modelling

3.687 The following section (1.4.1) is in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092].

3.688 Bats are most responsive to frequencies above 8 kHz which is towards the end of human hearing and therefore standard human noise assessments are not appropriate for the assessment of bat impacts (8.2.33 in 6.14 vol3 Chap2 ES Addendum Terrestrial Ecology & Ornithology Appendices 2.9A-2.9D (AS-208)). Barbastelle use passive listening and calls for hunting (rather than echolocation) and could be adversely affected by 8-20 kHz noise. It is therefore difficult to justify the use of human focussed modelling which is at a lower frequency (below 8 kHz) for the bat impact assessment (AS-033 14.13.105). Not only that, the human-focussed modelling uses average noise levels for the noise impact assessment which may not reflect true levels of impact compared to maximum noise levels (Volume 2 Main Development Site Chapter 11 Noise and Vibration Appendix 11B Construction Noise Assessment, App-204).

3.689 This approach has been used to assess impacts on Bridleway 19 and Stonewell Belt foraging and commuting areas. The noise modelling conducted, as outlined in this ES (Doc Ref. Book 6) (Vol 2, Chapter 11) APP-204 concludes that noise levels are unlikely to be significant at these locations (AS-033, Table 14.44). Unfortunately, this conclusion reflects the major limitations of the study in that it utilises noise estimates based on what humans can hear, using an average rather than maximums and hence may have little relevance in terms of impacts on bats.

3.690 Whilst it is recognised that the updated bat impact assessment (AS-208) has removed the reference to human noise modelling, it still draws the same conclusion of no impact from noise as the ES chapter AS-033. The modelling data from Volume 2 Main Development Site Chapter 11 Noise and Vibration APP-202 is the average noise levels for the two locations (Roundhouse for Upper Abbey bridleway and Ash Wood Cottage for Stonewell Belt) using modelling with a 5 metre acoustic screen. It is these data that concludes a non-significant impact. Critically, the noise and vibration chapter predicts night-time maximum noise levels at the Roundhouse could be as high as 67dB LAmax when the green rail route materials unloading and stockpiling is underway. In our view, this represents a potential significant adverse impact (References Chapter 11 Noise and Vibration Appendix 11B Construction Noise Assessment (APP-204) paragraph 2.5.5 and 2.21.5 and APP-202 paragraph 11.6.47 and 11.6.49) that appears to have not been fully recognised.

3.691 The limitations of the conclusions above are further compounded by the fact that the human-focussed modelling used noise levels at only 1.5m above the ground. This has only very limited application to a high-flying bat species such as barbastelle, which commonly roosts and forages between 10-20m above the ground.

3.692 Vibration from piling has been dismissed as an impact for bat roosts as no known roosts have been found close to the western platform or the SSSI crossing (8.2.3, updated bat impact assessment (AS-208)). We question how these conclusions can be reached definitively, given there has been no roost monitoring in the SSSI triangle and areas bordering where the piling will go.

Foraging

3.693 The updated bat impact assessment (AS-208) states, 'studies which have measured effects of noise >65dB have found that bats typically avoid habitats exposed to this level of noise' and 'noise levels exceeding 65dB have a negative impact on a broad spectrum of bat species' (8.2.51). The report then goes on to say (8.2.52) 'The evidence outlined above suggests that bats are able to quickly adapt when exposed to a range of 'high level' noise'. However, there is no evidence to support this conclusion in light on the evidence presented.

3.694 The ES chapter (Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology - Revision 2.0 [AS-033]) Table 14.43 and paragraph 14.13.100 show eight barbastelle foraging and commuting areas have the potential to experience noise levels above 65dB (at 8kHz and 10 metres height).

3.695 Specifically, along Kenton Hills, given the structure of the dense conifer plantation (in contrast to more naturalised deciduous wood) barbastelle will not forage as much in the actual woodland itself. They are more likely to forage along the edge of the wood, to the north, which is closer to the noise source and hence may be impacted by noise in this area. Consequently, any displacement forcing barbastelle to forage within the conifer plantation, is likely to result in sub-optimal feeding and potentially loss of condition for individuals. There appears to be no consideration of this.

3.696 The following section (from this point until the end of 1.4.2) is in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092].

3.697 There is also very little in the way of determination of noise impacts from the Green Rail depot along the edge of Kenton Hills. This is especially important given the recent proposed increases in night rail movements. It is likely that unloading of loads will create a significant noise source. If this is done at night, there is a risk to foraging bats along the fringes of the wood and bund. However, if unloading occurs during the day, there appears to be no assessment on the impacts on the barbastelle roosts close by.

3.698 Other areas of significant noise include the SSSI crossing, around the whole of Ash Wood and along Bridleway 19. Table 8.22 (updated bat impact assessment (AS-208)), for example, clearly shows Bridleway 19, which is proposed as a quiet/dark corridor for bats, to be over 65 dB at 22kHz, levels of which are very likely to disturb foraging. It is unclear how it is envisaged barbastelle will adapt and connectivity will be maintained in this instance.

3.699 The ES chapter AS-033 Table 14.43 says Goose Hill's eastern boundary is used as a commuting and foraging area, which will be subject to high frequency noise above 65dB. However, the updated bat impact assessment AS-208 Table 8.22 says noise levels will be below 65dB despite both referring to noise modelling data and results presented the Bat Mitigation Strategy (APP-252 and APP-253). It is unclear which is correct and hence it is impossible to determine noise impacts in this area.

Roosting

3.700 A threshold of 60 dB at 8kHz has been set for disturbance at roosts (APP-253 14.13.82). The noise assessment indicates (APP-253 14.13.96-97) there will be noise >65dB at 9 barbastelle roosts and multiple potential tree roosts during Phases 1 and 2 of construction. This equates to four active bat seasons (i.e. four potential generations), which is easily long enough for significant deleterious impacts on populations. Whilst it appears likely that most barbastelle roosts identified by radio-tracking are unaffected there is no information to say how many and therefore it appears hard to justify that adverse effect is not significant. Given there are significant levels of noise >65 dB and the threshold for disturbance at roosts is 60 dB, it is likely that many more potential roosts are affected.

3.701 Barbastelle move roosts regularly and this appears to be given for the reason there will be no significant impact. However, for a robust approach and conclusion, a noise impact on total roost resource should be assessed to determine overall impact but this does not appear to have been done.

3.702 From the evidence presented roosts R1, R2, R11, R150 within Kenton Hills will all be over the level of significant impact, as well as the five Ash Wood roosts. Given this, the conclusion that no noise license is required even though modelling clearly shows there will be impacts on roosts appears difficult to sustain (AS-208 8.3.64).

Location of Noise modelling

3.703 The following section (1.4.4) is in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021. [REP1-092].

3.704 Bats, especially barbastelle, roost and forage at up to 20m above ground. However, the noise modelling for bats was located at a maximum of 10m above ground level (APP-253 14.13.93). Furthermore, the noise modelling for human receptors which was used to determine impact, was at 1.5m (and 4.5m) above ground level. This further adds to the uncertainty of the conclusions: not only the use of inappropriate human noise modelling in the assessment but also the location of the noise modelling itself is not appropriate to determine impacts on barbastelle.

Timing of Noise Modelling

3.705 Clarification is required over hours of night-time noise modelling and assessment. Paragraph 1.2.2.2 of ES Vol 2 Chapter 11 Noise and Vibration and Appendix 11B Construction Noise Assessment (APP-204) explains that assessment scenarios have been defined separately for daytime (07:00–23:00hrs) and night-time (23:00–07:00hrs) construction. This means that "night-time" does not include all the dark hours throughout all of the active bat season. With full construction activities ongoing until 11pm, we are concerned regarding the potential for impacts of noisy activities during dark hours. Bats are active from around dusk to dawn which varies throughout the year and consequently modelling noise and assessing impacts from dusk to dawn may be more relevant. We would like clarification that all hours of bat activity are being assessed in a similar way.

3.706 It is uncertain from the data presented whether the noise modelling and assessment accounts for phase 5 of the construction and for significant overlap between construction phases. ES Vol. 2 Main Development Site Chapter 11 Noise and Vibration (APP-202) section 11.3 d) describes the scenarios assessed as part of the noise modelling and explains the types of activities planned to take place over the five construction phases during both the daytime and night-time construction

shifts. It is stated that construction Phase 1 (site establishment and preparation for earthworks) and Phase 5 (removal of temporary facilities/restoration of the land) will result in the highest noise levels, although it is also noted that the construction phases will overlap. It is therefore important that Phase 5 impacts are fully considered as are the times when the phases overlap.

Lighting

3.707 The following sections (1.5.1-1.5.4) are in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092].

Limited Spatial Coverage of Light Monitoring

3.708 Information provided on the lighting (Volume 2 Main Development Site Chapter 2 Description of Permanent Development Appendix 2B Lighting Management Plan APP-182 1.3.21) only uses three examples and does not include key areas used for roosting, foraging and commuting bats, especially for barbastelle. Key areas omitted include Ash Wood, areas around Upper Abbey Farm, southern end of Bridleway 19, Leiston Old Abbey, Fiscal Policy, much of Kenton Hills or the SSSI. Additionally, the examples that are used, are in areas of the development that will be inherently darker and hence not a true reflection of potential impact.

Bridleway 19

3.709 Bridleway 19, which is retained as a dark corridor, is most heavily used by commuting barbastelle towards the southern end, close to Fiscal Policy and Kenton Hills. This is also the area where there is most risk of impact by light and so it seems incongruous that the light monitoring and modelling is at the northern end of the bridleway, where there will be less light spill and evidence of lower use by barbastelle. Therefore, it is questionable if the light modelling in this area is of the accuracy needed to determine impact. Area 1 is located along the northern section of Bridleway 19, which will be some distance from the brightly lit access routes further to the south, areas which are also indicated from the bat survey data, to be more important for bats. In Area 1 where the modelling was undertaken, the accommodation light spill will be much lower than the access routes. Table 1.13 of APP-182 shows areas of very high lighting towards the lower end of Bridleway 19; Checkpoints (150 min lux), Gatehouse (200 min lux), Service areas (100 min lux) and Inspection areas (50 min lux) but there is no detail where these are located exactly in relation to the 'dark corridor' of Bridleway 19. Importantly, the light modelling does not appear to capture these. We would expect obvious light level contours showing them clearly, but this does not appear to be the case.

3.710 The actual light modelling along Bridleway 19 has focussed on calculation 2 (8.2.100 of AS-208 and APP-182 1.3.24), but this is at ground level. There was only one test carried out above ground level. Therefore, not only was the location of the tests in a suboptimal location, the testing was largely at ground level that has less ecological relevance to barbastelle that typically commute and forage at heights greater than 10 metres.

3.711 There are no details in terms of the width of each of the four severances along Bridleway 19. This is a significant omission given the bridleway is being championed as a dark corridor despite being severed by 2 haul roads, the main access road and a railway line.

Kenton Hills

3.712 Area 2 is along the northern part of Kenton Hills, however, it will be well away from the green railway drop down area and hence will also be subject to significantly lower levels of light spill. In

Table 1.8 of APP-182, in our view the most ecologically relevant calculations for the north side of Kenton Hills are calculations 6, 7 and 8 and these have a maximum lux of 42.7, 23.3 and 13.5. These are levels that without mitigation may have significant impacts on bats but there appears to be no consideration of these findings. Section 8.2.109 of AS-208 suggests calculation 5 is the most important for bats at Kenton Hills but the location of this monitoring is at ground level and appears to be behind the earth bund. Consequently, this site will have little relevance on commuting and foraging bats between 10-20 metres, such as barbastelle, as the light will not be buffered by the bund at this height.

Ash Wood

3.713 Section 8.2.97 (plate 8.61) of AS-208 indicates that Ash Wood has not been modelled for lighting impacts at all. Given Ash Wood is one of the most important areas of the estate for barbastelle and the fact that development activity will occur on three of the four sides of the wood, this appears to be a significant omission. For example, the contractor compound is directly to the south of Ash Wood, likely with a high level of light and disturbance noise, plus close proximity of the haulage road. None of the modelling appears to reflect this (AS-208).

SSSI

3.714 The lighting assessment for the SSSI is all at ground level (8.2.115 of AS-208) and hence only has limited relevance in determining possible impacts on barbastelle foraging and commuting, or on roosts in the area. This coupled with the fact there has been no monitoring for bats in the SSSI, or even search for bat roosts, means there is no way to determine impacts.

3.715 Despite this, the Lighting Table 1.9 of APP-182 still shows there will be modelled light spill onto the SSSI of up to 18.4 lux, but there is no explanation as to the possible impact this will have on bats within the SSSI. If the modelling was done at a more ecologically relevant height for barbastelle, the light levels may indicate significant impact on this species. Unfortunately, given there has been no data collection for barbastelle within the SSSI, either for roosts or commuting and forage, we would be unable to determine what the impact would be, even if we had these data.

Mitigation

Uncertainty of Effectiveness due to Task Specific Lighting

3.716 It remains unclear how, in practical terms, unacceptable levels of light (and noise) will be defined and mitigated during construction. There appear to be potential conflicts between health and safety and further controls being implemented. We have major concerns over the lack of detail on task-specific lighting during construction and how this will be mitigated i.e. controlled, practically speaking on-the-ground on a day-to-day basis and at present there is nothing presented that could be easily adapted to provide the basis for a Working Method Statement for an Ecological Clerk of Works.

3.717 There is considerable uncertainty over how effective the mitigation proposed will be in practice. In section 8.2.89 of AS-208, the lighting mitigation states that if a dark corridor is dissected by a road there would be a gap in the lighting of 30 metres but only ‘if safe to do so’. There is no information which areas are ‘safe’ to include and which are not. Section 1.3.17 of APP-182 states, ‘where foraging routes and flight paths interconnect over the temporary construction area, where reasonably practicable these connecting areas shall be left dark’. It is left unclear as to what ‘reasonably practicable’ means in reality. As a result, it is impossible to determine the actual level of impact and hence how effective the dark corridors will actually be.

3.718 In section 1.4.16 of APP-182, it states that Sizewell Marshes SSSI should be kept as dark as 'reasonably possible' but as with elsewhere in the report, there is no way of knowing what this actually means. More information is needed to determine impact on the SSSI not only bordering the culvert but also on the western edge of the SSSI, bordering the main platform.

3.719 There is very little work and consideration over the type of lighting. It is not just the level of Lux that is important, but whether it is blue-white or yellow-white light. Differences between tone of light can have a considerable difference in levels of disturbance. Volume 2 Appendix 2B Lighting Management Plan for Construction and Operational Sites (APP-182) Section 1.2.59 proposes that the use of UV lighting, which the scientific literature suggests has a bigger impact on bats, will kept to a minimum 'where practicable'. However, there is no supporting evidence that shows there has been any analysis of what the impacts would be where this is not be practicable and hence the reader is unable to determine impact.

3.720 Section 1.2.69 of APP-182 illustrates the use of LED could increase white source light and white source light may increase impacts on bats, but there is no detail on how LED lighting will be kept to a minimum in areas of high bat activity, or how this lighting will be changed to less impactful lighting. Section 1.4.18 does mention that the use of lights with no UV, warm colour and tuneable LED will be used, 'where reasonably practicable'. Again, there is no way for the reader to determine what this actually means and hence determine final impact.

3.721 The borrow pit and stockpiles will have occasional task specific lighting (Table 1.6 of APP-182) but there are no details for how long, how bright and what type of lighting they will be and so there is no way to judge impact. Task specific lighting 'may be switched off when required' but there is no commitment or detail when or how this will happen in practice.

3.722 In essence, the lighting strategy needs to define exactly what is meant by '*where reasonably practical*' in all circumstances in which this, or similar, terms are used. Without this clarification, it is impossible to determine impact. There is also a need to be clear how, operationally, this will be done. This is especially important around controlling task-specific lighting, close to key features such as Ash Wood.

3.723 The strategy then needs to inform an operational lighting plans for construction that needs to include a clear and realistic worst-case scenario that can be compared to the planned mitigation. The lighting plans should follow best practice guidance for bats.

Limited Buffering of key Areas

3.724 Paragraph 1.3.19 of APP-182 also states that if lighting runs parallel to commuting routes and flightpaths (such as along the dark corridor of Bridleway 19 and around Ash Wood), then the buffer will only be 10m. Figure 2B.1 puts much of Ash Wood surrounded by Zone B (as is much of the SSSI). Bridleway 19 is crossed in 3 places by Zone B, areas which appear to include Gatehouses and Checkpoints and therefore subject to significant lighting. In areas of high intensity lighting, which appear to be very close to Bridleway 19, it is very unlikely a buffer of 10 metres will be enough to prevent significant fragmentation of what is supposed to be a dark corridor. A buffer of only 10 metres around Ash Wood may also result in significant impacts on roosting and emergence.

3.725 Pill Box field still appears in plans with an apparent footpath directly across the SSSI. What are the impacts from lighting on the SSSI (in addition to the direct land take from the SSSI)? No details are given.

3.726 Across the site, it would be helpful if there were figures showing the retained dark corridors with light modelling overlaid (from more areas and at heights that are ecologically relevant to barbastelle) pre and post mitigation. This would also help to understand light spill into the SSSI.

In Combination and Synergistic Impacts

3.727 Table 10.1 within the 6.14 vol3 Chap2 ES Addendum Terrestrial Ecology & Ornithology Appendices 2.9A-2.9D (AS-208), assesses in-combination effects for light and noise but there appears to be no in-combination assessment for light, noise and physical fragmentation (e.g. felling of Goose Hill woodland) all together. This appears to be a significant omission given the conclusion that there is likely to be a significant impact on a nationally important population of barbastelle bats and a county important Natterer's population, as a result of habitat fragmentation as an effect, when considered in isolation. However, in addition to this, there is considerable uncertainty relating to the impact from light and noise which may increase the residual impact already noted, due to synergistic effects.

3.728 Furthermore, there has been no attempt to understand the interrelationship between noise and light, especially the likely direct correlation between task specific intensive lighting and subsequent noise. The only reference to this is 8.2.125 in AS-208, 'when increased levels of task-specific lighting do correlate with higher noise levels, these events are likely to be short in duration relative to the construction period'. Given the length of the construction period is likely to be well over a decade, this is likely to be correct, but this still means the impact could span several years and across multiple breeding seasons.

Goose Hill

Barbastelle roosts within Goose Hill

3.729 The evidence presented suggests that Goose Hill may contain barbastelle roosts that have not yet been located, that will be lost due to felling. There were notable levels of barbastelle activity from 23-24th June 2020 between Hilltop Covert and Goose Hill. Based on these data, the back tracking report concludes the most likely barbastelle roost locations are in Kenton Hills or to the north, at Minsmere, with no justification as to how these conclusions were reached. However, data illustrated in Plot 4 (page 40) from the static survey (AS-037) suggests no early emergence of barbastelle from detectors around Kenton Hills but a number of early emergence records from a detector (MS20) on the southern edge of Goose Hill. These early emergence records strongly indicate roost sites either within Goose Hill or at least very close to the edge. No attempt has been made to investigate these findings further in the 2020 static or back-tracking surveys. Therefore, there remains considerable uncertainty over the conclusion that all the barbastelle roosts are within Kenton Hills and none within Goose Hill.

3.730 Section 5.3.7 of APP-253 says that Goose Hill has 'minimal roosting resource for bats', although it is hard to sustain that position given there are 7 trees with High and 104 with Moderate roost potential, all of which will be felled. Furthermore and perhaps most importantly, Figure 2.9B.1 of the updated bat impact assessment (AS-208) shows a barbastelle roost in Hilltop Covert, which is part of the western block of Goose Hill. It is unclear if this will be lost.

Importance of Goose Hill for Commuting and Foraging

3.731 There is a confirmed loss of barbastelle commuting and foraging habitat through Goose Hill including activity at MS11, identified in section 5.3.16 (6.14 AS-208 Vol. 3 Ch. 2 ES Addendum Terrestrial Ecology & Ornithology Appendices 2.9A-2.9D). This suggests consistent barbastelle activity within the development footprint, especially around and within Goose Hill. Section 5.3.20 of AS-208 also observed that some of the most important areas for bats included the northern

edge of Goose Hill woodland bordering the SSSI crossing and the southern edge of Goose Hill adjacent to the SSSI 'triangle'. Further, in section 5.3.27 there is an acknowledgement that Goose Hill is the only confirmed barbastelle commuting route on the estate, which will be lost. Within the Backtracking Report (AS-037), Table 5 notes there were notable levels of barbastelle foraging at TM464 645502. This is within Goose Hill and again suggests the importance of this area which will be directly lost under the development.

3.732 In Section 5.3.28 of the updated bat impact assessment (AS-208), four Natterer's bats were radio tracked and two were found to be using Goose Hill exclusively and a further individual foraging exclusively within Goose Hill and Hilltop Covert. Table 8.24 states that Goose Hill is of particular value to Natterer's bats 'later in the season'. Again, the importance of the total loss of Goose Hill to the development does not appear to have been fully recognised or the impact mitigated.

Importance of Goose Hill for breeding female and juvenile barbastelle

3.733 The Baseline surveys show Goose Hill as important for feeding resource for breeding females and juveniles, with their restricted range and flying ability. Table 8.18 (AS-208) indicates that the 2011 radio tracking work concluded that Goose Hill was a key foraging area for barbastelle on the estate during pre-lactation period and again in 2014, was important for females post breeding. Table 8.20 states that a pre-lactation commuting route from Ash Wood through to Goose Hill has been identified for barbastelle.

3.734 Due to the small hunting ranges of breeding females, Goose Hill is likely to be close to, or even include, a maternity site that will be lost. This has not been fully recognised and there remains a significant risk that there is both a direct loss of a barbastelle maternity colony and subsequent impact from the development on foraging for poor flying pregnant females and juveniles. Due to their limited ranges, it is unclear whether these bats would be able to use the SSSI crossing and hence may face significant impact. Despite this, there has been no work to understand what percentage of home range of females and juveniles will be lost and what this impact would look like. Whilst the SSSI crossing is likely to be used for bats (assuming the lighting plan is adequate) there is still likely to be a significant effect on barbastelle even if they use the culvert, due to an increased distance required to reach foraging areas after Goose Hill has been felled. It is not clear either how this residual effect will be compensated for, or how this effect will specifically impact poor flying juvenile and lactating female barbastelle using Goose Hill previously.

Sizewell Link Road (SLR)

Importance of the SLR to bats

3.735 We are concerned that the importance of the SLR has been underestimated, the bat usage of which is described in the report as 'very low levels'. However, Detector 7 at this location Volume 6 Sizewell Link Road Chapter 7 Terrestrial Ecology and Ornithology Appendix 7A Ecological Baseline and Method Statements (APP-462) Table 1.19) has the second highest encounter rate across the development site, including across the Main Development Site. Further evidence highlighting the importance of the SLR for rare bats is illustrated by the fact that of the 64 static recordings, 50 registered barbastelle. There are no follow up data to determine if this is simply a few bats moving up and down, or a significant directional movement (for example, by using thermal imagery with static recording) but it does suggest that the conclusion of very low levels is incorrect.

3.736 Furthermore, whilst the importance of the SLR is underestimated based on current use, its importance for barbastelle is only likely to increase during the construction if displacement occurs.

SLR Mitigation

3.737 Given the likely importance of the SLR area during the development period, we believe that there needs to be more mitigation provided, especially in relation to the provision of bat boxes suitable for barbastelle. There is also concern over effectiveness of the current mitigation using standard trees; these will need to be set back from the road, in dark areas and will be to be planted at a height that aids commuting.

3.738 Furthermore, it is not clear what work has been done on determining impact resulting from road collisions along the SLR and what the combined effect might be with other stressors such as light and noise.

3.739 The design and location of road crossing points for bats should follow best practice (Berthinussen and Altringham, 2012[382] and 2015[383]). Data from studies on mitigation for bats and roads showed that bats used underpass located on a pre-construction commuting route that allowed bats to pass without changing flight height or direction. Attempts to divert bats from their original commuting routes were unsuccessful.

3.740 Table 1.2 of the bat mitigation strategy (APP-252) notes these requirements relate to the provision of safe crossing points over linear developments (road, rail) and are of less relevance to the provision of a protected corridor through a 'hostile' landscape. We consider these requirements are directly relevant to the SLR.

Mitigation

3.741 The following sections (1.9.1-1.9.3) are in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092].

Central Dark Corridor (CDC)

3.742 The Addendum appendices rightly points out that barbastelle, 'would experience direct habitat loss and fragmentation of commuting routes and be impacted by increased noise and lighting'. (6.1.8, 6.14 vol3 Chap2 ES Addendum Terrestrial Ecology & Ornithology Appendices 2.9A-2.9D AS-208). On this basis, we believe a CDC which is unimpacted by light and noise, is established through the centre of the site. This may offset, to some extent, the loss of Goose Hill.

Use of SSSI crossing as Mitigation

3.743 Whilst the wider culvert over the SSSI is an improvement on the original design, it cannot be said to 'promote connectivity' (AS-208 8.4.35) beyond what is already there, as the baseline is of SSSI quality and likely to be well used by commuting bats anyway. Given no monitoring has been carried out within the SSSI triangle, there is no current baseline from which to judge but despite this, it seems strange that the crossing and loss of SSSI is being proposed as mitigation for bat connectivity.

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382 Berthinussen, A. & Altringham, J. (2012) Do bat gantries and underpasses help bats cross roads safely? PLoS ONE 7(6): e38775. doi:10.1371/ journal.pone.0038775

383 Berthinussen, A. & Altringham, J. (2015) Development of a cost effective method for monitoring the effectiveness of mitigation for bats crossing linear transport infrastructure. University of Leeds/DEFRA.

Lower Abbey Farm Bat House

3.744 The provision of a new bat house at Lower Abbey Farm is a welcome further improvement to the mitigation package (2.2.112 in 6.14 6. Environmental Statement Volume 1 Addendum Chapter 2 MDS AS-181). However, the relative lack of wider mitigation to ensure the location within the wider development itself will remain attractive to bats, places considerable uncertainty over whether the barn will be used, or its potential maximised. The surrounding area is very likely to be severely impacted by light and noise and so we question the mitigation value of the bat house in its current location unless a more robust plan for lighting and noise mitigation is provided, especially along the southern end of Bridleway 19 where there is likely to be significant light and noise. Furthermore, if the bat house is proposed to mitigate the loss of the barn at Upper Abbey Farm (this is not clear in the documents), it is unlikely to provide suitable mitigation for the multiple species roost within the barn that will be lost at Upper Abbey Farm. To achieve this, a larger and more complex design will be needed to achieve the differences in access, light, temperature and humidity required by multiple species currently found in the existing barn. Finally, it is a missed opportunity not to provide accessible, sensitively designed subterranean spaces below the house that could act as hibernacula. This may be especially important to barbastelle given there is no clarity where the bats go to hibernate at times of very cold weather and hence there is currently considerable doubt as to whether the development will prevent access to historic hibernation sites. The provision of a further hibernation sites at Lower Abbey Farm would help to mitigate this risk.

Light and Noise Mitigation

3.745 There remain a number of areas that have high levels of barbastelle activity that appear to have insufficient mitigation in terms of light and sound. As discussed previously, the lack of monitoring in many of these areas means there is considerable uncertainty of impact. Specifically, mitigation appears likely to be lacking along the Bridleway 19 corridor, Fiscal Policy, along three sides of Ash Wood, Black Walks and the link to Minsmere and around the SSSI crossing and to the west of the main platform.

3.746 Triggers for noise and lighting remain undefined, especially in relation to task-specific lighting. Until there is some indication of how noise and lighting impacts will be practically controlled, it is difficult to determine impact and effectiveness of mitigation.

3.747 The following sections (from here until 1.9.4) are in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092].

3.748 In the bat mitigation strategy, APP-253, Fig 14C1A.5 suggests that the earth bund mitigation to the north of Kenton Hills covers four of the barbastelle roosts within the red line boundary. It is not clear which side of the roosts the bund will be. The figure also shows foraging along the northern edge of Kenton Hills boundary actually within the conifer plantation. However, the foraging is much more likely to be directly under the bund footprint i.e. along the edge of the oak trees, than within the conifer plantation. So, the bund in its current location appears to directly impact on barbastelle roosts by covering the oak trees.

3.749 As a general, but important comment, the proposed mitigation bunding across the site, such as at Kenton Hills and Ash Wood, will not buffer from light and noise impacts due to the fact the barbastelle will be naturally foraging over the top of the bund; the proposed bunds are five metres tall, whilst barbastelle are known to forage and commute in excess of ten metres.

Concern over Assumptions of benefits to bats from other Mitigation Sources

3.750 We are concerned that the provision of Aldhurst Farm, the marsh harrier improvement area and Studio Field Complex (updated bat impact assessment, AS-208 8.5.32) are mentioned in relation to wider mitigation, especially given the clear loss of home ranges for some barbastelle are found to be of medium to high magnitude significance (8.3.29). It is unlikely these areas will provide adequate compensation for this residual impact due to: a) difficulty in accessing these sites due to loss of overall connectivity; b) poor forage habitat of these areas for barbastelle – much of these areas will consist of open, dry grassland with low levels of invertebrate prey and it is noticeable that overall the Environmental Statement is too simplistic as it assumes overall biodiversity gain with changing land use on the estate will benefit bats. This is a significant assumption as large losses of woodland may not be offset by acid grassland creation, which is not prime bat foraging habitat compared to woodland edge; c) the sensitivity of pregnant females and juvenile barbastelle around Goose Hill and Fiscal Policy and their small home ranges, being unable to access these sites due to distance effect; d) lack of evidence as to what the baseline was in these areas – there is no evidence these new areas will provide any better foraging habitat compared to what was already there.

Bat Box Mitigation Strategy

3.751 Much emphasis has been placed on the provision of bat boxes as part of the mitigation and whilst barbastelle will use bat boxes, they are not an effective compensation for the wholesale loss of trees, given barbastelle switch trees on a regular basis. Barbastelle also switch bat boxes regularly and many boxes are required, not just to replace known roosts, as known roosts are likely to be a fraction of the actual roosts lost. For example, 74 medium and high potential tree roosts in Ash Wood are likely to be impacted by noise and light in addition to the 111 medium and high potential roosts in Goose Hill that will be lost entirely. Furthermore, there has been no work done on roost potential within the SSSI triangle or to the west of the platform. Consequently, the current provision of boxes appears to be very low. The Backtracking Report records that 'potential roosts have been noted but not identified for barbastelle in Goose Hill and Broom Covert' (AS-037, 2.3.4). Again, this suggests the box replacement mitigation is not enough to offset the numbers of potential roosts being lost, especially within Goose Hill and other areas such as the SSSI triangle and to the west of the platform and SSSI edge. There is an urgent need to map areas that can take more boxes, or areas which currently have limited opportunities for roosting bats currently, as places for suitable further mitigation. In essence, there needs to be a much greater commitment for box provision over a wider area. We request details of the design of bat boxes installed on the estate and if there is any evidence barbastelle have used them. Greenaway and Hill (2004)[384] note special boxes 'designed to offer narrow gaps with a long access length appear much more attractive to them' (section 3.6).

3.752 It is also worth noting that bat boxes should not be the only approach to roosting provision. For example, it is notable that a single dead Scot's pine on the eastern edge of Kenton Hills is the most important roost found for biggest range of species. It would also make sense to veteranise a number of pine trees in suitable locations (with features added to enhance their desirability), between 20-30, across the estate.

3.753 There should be key foraging areas close to retained and newly provided roosting resource and along bat flightlines. Greenaway (2004) note 'When too many foraging opportunities are lost along a flightline, the bats will cease to use it, even if its connectivity is maintained. Increasing

[384] Greenaway, F. & Hill, D. (2004) Woodland management advice for Bechstein's bat and barbastelle bat. English Nature Research Report No. 658, English Nature, Peterborough

foraging opportunities along the sections of flightline used in early evening lowers the requirement for long distance commuting in poor weather.'

3.754 Greenaway and Hill (2004)[385] note 'abundant productive forage close to the roost' when juvenile bats first start fly is 'key to the success of any bat colony' (section 3.4) and suggest management plans for nursery roosting sites should include areas of five to ten times the area that bats are presently using to ensure a continuing supply of roost sites for barbastelle bat colonies into the future (section 3.5).

Deer Management

3.755 The following sections (1.9.7) is in addition to observations raised in the Joint Local Impact Report - Appendix 2.3 BSG Ecology Review of Bat Impact Assessment October 2020 [REP1-091] and Appendix 2.4 BSG Ecology Review of Bat Impact Assessment: Second Review March 2021 [REP1-092].

3.756 It is apparent that the control of deer has ceased on the estate. Why this is the case is currently unclear, but increased deer numbers over time will increase pressure on the woodland edge as well as the developing scrub/edge habitats in the arable reversion. It is highly likely over time this would further reduce the quality of foraging habitat for bats. A planned and sustained deer control policy would go some way to retain high quality habitat for foraging bats over the longer term.

Lack of Compensation for Residual Impact

3.757 Perhaps most importantly, the updated bat impact assessment (AS-208) notes there remains a residual Moderate Adverse Effect on barbastelle that is Significant even with the new connectivity corridor, wider culvert, bat box provision and bat house. Despite this, there appears to be no attempt to provide any compensation (8.3.47).

Conclusion

3.758 We are concerned that a lack of abundance data, a low coverage of data gathering from across the site and limited data gathered from key times of year, has resulted in a poor-quality baseline, from which to draw robust conclusions. This has been compounded by analysis focussing on the entire bat population rather than a focus on the rarer species found on the estate.

3.759 As a result, the mitigation provided is unlikely to be enough. Where mitigation has been provided, due to a lack of detail regarding specific working practices, there is considerable uncertainty as to whether this will be effective. Due to inherent uncertainty over noise and light impacts on barbastelle, there is a lack of a precautionary approach around key areas of the site.

3.760 Having concluded significant effect on barbastelle bat, there is no attempt to compensate for residual effect.

3.761 There is no structured monitoring plan in place to determine future population trends, from which to respond to potential future impacts.

3.762 If barbastelle continue to roost within the EDF estate, there is likely to be a population level effect as a result of displacement of females from foraging habitats, due to the construction area representing a partial barrier to movement.

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[385] Greenaway, F. & Hill, D. (2004) Woodland management advice for Bechstein's bat and barbastelle bat. English Nature Research Report No. 658, English Nature, Peterborough

Natterjack Toads

Summary

3.763 This natterjack section is informed by the report; Amphibian and Reptile Conservation (2020). Sizewell C Project: comments on natterjack toad mitigation proposals. Report to Suffolk Wildlife Trust. ARC report. ARC, Bournemouth. The report is attached as an appendix. We respectfully request that the Examining Authority review the Report in full as text below is just a summary of the points covered within that Report.

3.764 The main findings of that Report are:

- The framing of impacts as "temporary" when they would in fact last for three natterjack toad generations is a notable misinterpretation of the potential impact on the viability of the population;
- There is insufficient reassurance that habitat creation and management proposals would generate and maintain a resilient population of natterjack toads; and
- The proposals do not adequately take into account the ecology and local conservation status of the population.

Dataset

3.765 The data are generally well presented and largely sufficient for this type of impact assessment with the notable exception of surveys in terrestrial habitat.

3.766 We agree with the conclusions about the status of the natterjack toad population whereby the report adequately summarises the long-term monitoring data and the suggestion of a population size of around 30 adults is reasonable[386]

3.767 However, it would have been appropriate to caveat this with a note about the fact that surveying/monitoring effort has not been consistent and intensive enough for a precise and reliable population estimate. Although we do not think that the use of terrestrial habitat in Retsom's Field and surrounding areas has been well described.

3.768 Additionally, there are concerning comments on the ecology of natterjack that raises the question over the level of expertise the author has with regard natterjack ecology specifically. There is an implication that natterjack toads might be considered poor swimmers which needs further clarification [387] as well as the intention of the comment about natterjack toads not being found during reptile surveys [388] Both require further context to further understand what is meant by these comments and how this has influenced the design of the mitigation.

3.769 We note that there is an error in the section describing the legislation applying to natterjack toads.[389]. The text indicates that that killing, injuring or taking natterjack toads is an offence under the Wildlife and Countryside Act 1981 (as amended), but this is not the case. Most provisions of Section 9 of that Act do not apply to natterjack toads, with the main protection for these species

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386 ES V2 Ch14 Terrestrial Ecology Ornithology Appx 14A5 Amphibians - pg 13, para 1.2.25 (doc pg: 9) (APP-233); pg 140 para 14.10.5 (doc pg: 134) (AS-033); ES 2 Ch14 Appx 14C Protected Species – pdf pg 330, para 1.2.15 and App 14C7A Natterjack Toad Mitigation Strategy - pg 9 (APP -252)

387 ES-V2 Ch14_Appx 14C_Protected_Species, pg 354, para 1.1.12; App 14C7A Natterjack Toad Mitigation Strategy pg 23 (APP-252); and ES Vol 2, Ch14, pg 144, para 14.10.22 (doc pg: 138) (AS-033)

388 ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species, pg 358, para 1.3.5; and App 14C7A Natterjack Toad Mitigation Strategy pg 27 (APP-252)

389 ES V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 350, para 1.1.2 (doc pg: ES V2 C14 App 14C7A Natterjack Toad Mitigation Strategy - pg 19) (APP-252)

being within the Conservation of Habitats and Species Regulations 2017 (as amended). Again, this raises the question of the depth of knowledge of the author specific to natterjack toad.

Future Monitoring

3.770 The monitoring proposals are inadequate. It is not explained precisely what would be monitored or the methods, effort, timing, responsibility for delivery, use of monitoring results and sharing of data.

3.771 In a clear example of the current limitation with regard future monitoring, there is mention of monitoring being delivered annually during Water Management Zone (WMZ) operation and biannually over a 6 year period afterwards[390], but in order to obtain a mark recapture estimate we would expect annual monitoring.

3.772 In terms of monitoring objectives, we suggest that the following should be covered:

- Assessing the efficacy of mitigation measures put in place to reduce harm to natterjack toads during WMZ construction and operation;
- Assessing the efficacy of habitat creation and enhancement measures in the short and long term.

3.773 In terms of field survey methods used, we suggest two broad categories are included:

(1) Standard monitoring methods for this species: counts of spawn strings laid, assessment of metamorph production, counts of adults in ponds during the breeding season, assessment of habitat quality;

(2) A finer resolution of population status assessment: an estimate of adult population size through repeat surveys of adults and applying standard mark-recapture statistics, using natural markings to identify individuals (see, e.g. Smith et al 2018[391]; McMurray undated[392]).

3.774 Routine monitoring during the construction phase should be undertaken annually with at least 7 survey visits per year, as recommended for standard natterjack toad monitoring (e.g. see guidance on ARC's website at: https://www.arc-trust.org/targeted-surveys-for-natterjack-toads). During the capture phase, we would expect active capture by searching at night with a powerful torch, rather than a reliance on pitfall traps which appears to be the case with current proposals.

3.775 Monitoring should be undertaken annually for at least 6 years following completion of WMZ operation, in order to detect any residual impacts of the development and to trigger remedial actions. Monitoring should cover all the existing ponds (N1-4) and newly created ponds. There should be a clear statement of investigations and potential remedial actions should monitoring detect problems such as a failed pond liner. These investigations and remedial actions should be costed and secure resourcing plans put in place.

3.776 Assessment of habitat quality should include a note of management operations (e.g. grazing) and natural factors (e.g. rabbit activity, weather) likely to be influencing habitat condition. Monitoring should be carried out by personnel meeting standard criteria (CIEEM, 2013)[393]. Monitoring results

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[390] ES_V2 Ch14 Terrestrial Ecology Ornithology Appx14C Protected Species.pdf – pdf pg 345, para 1.4.12 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 14) (APP-252)

[391] Smith, S., Young, T., & Skydmore, D. (2018) Effectiveness of the field identification of individual natterjack toads (*Epidalea calamita*) using comparisons of dorsal features through citizen science. Herpetological Journal, 28(1)

[392] McMurray, C (undated) Estimating the natterjack toad (*Epidalea calamita*) population size at RSPB Mersehead reserve using a novel identification technique. [Unpublished note of placement with RSPB, undertaken in 2016]

[393] CIEEM (2013) Competencies for Species Survey: Natterjack Toad. Technical Guidance Series. CIEEM, Winchester

should be shared with those with an interest or involvement in managing the natterjack toad population, and with the national monitoring programme operated by ARC.

Potential Impacts

3.777 There is a high likelihood that rabbit grazing would be reduced or possibly re-distributed by WMZ construction activities arising from noise and lighting. Rabbits play a crucial role locally in the ongoing maintenance of a low sward, important for natterjack and this could be compromised. Displacement of rabbits would also reduce the provision of bare ground and underground voids through rabbit warren excavations, leading to a reduction in refuge availability. It appears this impact has not been recognised in the assessment[394].

3.778 There is the possibility that the WMZ construction and operation would result in significant changes to the hydrology of Retsom's Field and surrounding areas. If the field were to be either inundated, or conversely made significantly drier by the works, this would likely have an adverse effect on the natterjack toad population. We do not agree with the simplistic conclusion[395]

3.779 That hydrology is not an issue simply because the existing breeding pond (N1) is lined and the substrate is sandy. Were the Field to be inundated during the winter, this would have a significant impact on the survival rates of hibernating natterjack. Conversely, if the field were to become significantly drier, this might impact the suitability of the substrate for natterjack toads, which require some water content in the substrate to avoid desiccation when taking refuge. Hydrological changes might also affect prey availability. In addition, there is the possibility that hydrological changes might affect pond N4, which is unlined.

3.780 We believe the impact of the "temporary" loss of terrestrial habitat during WMZ construction and operation has been significantly under-estimated and is not adequately explained[396]. Natterjack toads are highly terrestrial, with most adults spending the majority of time on land and only a small fraction of the year in aquatic habitat; post-metamorphic immature individuals rarely, if ever, enter aquatic habitat. The exclusion of 3.5 ha of good quality terrestrial habitat close to the breeding pond for 3 generations could have a noticeable impact on the population through reduced foraging and refuge opportunities. The small size of the population, and its isolation, may exacerbate any negative effects.

Mitigation

3.781 The mitigation proposals indicate at several points that conventions used in great crested newt mitigation have simply been applied to natterjack toads, despite the fact that there are marked differences in the ecology of the two species. Again, this raises a question over the expertise of the author specific to natterjack toad. The mitigation proposals do not take into account some important ecological characters of natterjack toads, nor have they been sufficiently adjusted to account for this particular population.

3.782 There is a disconnect between the section of the EDF proposals which discusses natterjack toad ecology[397], and the proposals for mitigation and compensation: the former do not appear to have sufficiently informed the latter. For example, it is unlikely that 30 days of capture effort would be

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394 ES V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 341-342, para 1.3.1-1.3.10 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 10-11) (APP-252)

395 ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 342, para 1.3.8 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 11) (APP-252)

396 ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 341, para 1.3.6 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 10) (APP-252)

397 ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 353 App 14C.7A.3 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 22) (APP-252)

sufficient to adequately deplete the population in the fenced area, and the apparent reliance on pitfall trapping is insufficient[398].

3.783 There appears to be no mitigation in place to ensure the local rabbit population, who's activity is critical to create the right conditions for natterjack and might be otherwise displaced by the Water Management Zone construction activity, are not displaced by noise and light. More thought needs to be given to encouraging rabbit activity to create burrows and open, bare ground habitat.

3.784 There needs to be a thorough and comprehensive hydrological study on whether the WMZ might impact on the pond, hibernation site and surrounding area to understand if this might affect the natterjack population. If impacts are possible, mitigation needs to be sought to protect the hibernation site and wider foraging areas.

3.785 The mitigation currently fails to consider the potential impact on the 'temporary' loss of terrestrial habitat so close to the breeding pond over the course of three generations. Far more mitigation, specific to the ecological requirements of natterjack, is required to negate the likely impact on the wider terrestrial environment.

3.786 More generally, the particular vulnerabilities of this population, in conservation status terms, are not sufficiently addressed. This population is currently based on a single breeding pond, and if that were compromised the population would perish, resulting in a loss of range of significance at a much wider spatial scale.

Compensation

3.787 The proposed compensation for the loss of terrestrial habitats is inadequate.

3.788 The compensation proposals should be prefaced by a clear ambition. We would like to see a commitment to securing a resilient population in the long term, based on the availability of several breeding ponds set in good quality terrestrial habitat, with a local distribution expansion to the north.

3.789 A cluster of ponds of slightly differing profiles would be more appropriate than a single pond. This would increase the chance that in any one year suitable breeding habitat would be available, buffering against changes in environmental conditions. Habitat creation should involve a network of at least four new ponds, with slightly differing profiles and designed to provide good breeding habitat for natterjack toads. These should be sited in Retsom's Field, around the location currently proposed for N5, providing breeding opportunities between the existing breeding pond and N4.

3.790 Presently the proposal is that a single new pond would be created, and that this would effectively generate new terrestrial habitat. However, the terrestrial habitat in the field where the pond is proposed to the north is currently poor for natterjack toads, largely because the vegetation is rank. There is no meaningful mention of how the new breeding pond and terrestrial habitat would be maintained in suitable condition for natterjack toads after construction. Therefore, there is a lack of confidence that sufficient terrestrial habitat would in fact be created and maintained.

3.791 This is further illustrated by the fact that N3 to the north has not yet been colonised; this is likely because of the small source population, distance to that source but also because of the poor surrounding terrestrial habitat. These issues would not be sufficiently addressed by the EDF proposals for the new pond, N5. However, it is in principle a sensible idea to create a "stepping

[398] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 346-347, Table 1.3 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 15-16) (APP-252)

stone" pond between N1 and N4 but more needs to be done to ensure the surrounding habitat is suitable.

3.792 In setting out any plans for new lined ponds, it has not been recognised that they have a finite lifespan, typically 20 years, though this can be much shorter depending on materials and local conditions. Therefore, a programme of pond restoration and creation needs to be in place.

3.793 More detail on the specifics of the management of the surrounding area is required to understand how the landscape surrounding the ponds, from N1 up to N4 will be improved. For example, type and numbers of grazing animals. In addition, the newly created ponds will need to be well designed, with shallow margins, and be maintained so that aquatic vegetation, competitors and predators do not proliferate. In effect, measures have not been suggested as to how there will be continued provision of suitable breeding habitat.

3.794 The proposed linear feature[399] is unlikely to have any effect on connectivity. Natterjack toads are more likely to disperse over flat grassland, were it to be of a suitable sward, than to use a linear mound for dispersal; it is a feature that would be more suitable to other species of amphibians. The feature would, however, create terrestrial refuge opportunities if well constructed but improvements to connectivity would need to supplement this feature.

3.795 Overall, the scale of the proposed compensation habitat is inadequate given the potential impacts. Given the significance of the population, its vulnerability in terms of size and isolation, and the risks posed by the development, it would be appropriate to create a larger extent of quality habitat, with the aim of extending the local distribution of the species and increasing its viability. That could buffer against any residual impacts caused by the construction and operational phase. The natterjack toad is a pioneer species and populations can decline rapidly if habitat (both terrestrial and aquatic) conditions are not adequately maintained. The EDF proposals do not give reassurance that there would be measures in place to ensure a sufficient extent and quality of habitat in the long term.

Landscape strategy

Summary

3.796 We are concerned the Application will have a significant visual impact on visitor experience and the views available at RSPB Minsmere and the landscape strategy lacks measures to minimise and mitigate for this potential impact.

3.797 We are concerned the landscape strategy lacks sufficient details of baseline information, ecological objectives for habitats, species and ecological connectivity, habitat creation and management, robust monitoring and further interventions to be implemented if required and legal means of securing this throughout the lifetime of the development. We consider the Application should be focusing on landscape scale net gains and providing a functional ecological mosaic/ecotone and improving the actual ecological functionality of the surrounding area.

[399] [SZC_Bk6_ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species.pdf – pdf pg 343, para 1.4.8 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 13) (APP-252)]

Introduction

3.798 We have engaged with the Applicant throughout the pre-application and pre-examination period. We reviewed the DCO application and expressed our concerns in our relevant representations [RR-1059 (RSPB)[400] and RR-1180 (SWT)[401] and our Change Application responses].

3.799 We outline our current position below.

Main Development Site

RSPB Minsmere visitor experience

3.800 As noted in our Land Matters section below the Application will have a significant visual impact on visitor experience and the views available at RSPB Minsmere. We do not agree with the Applicant's position that, impacts on the landscape and visual effects are only localised, across the AONB. For example in the Landscape and Visual Impact Assessment[402], the Applicant does recognise this impact but states it would be localised across the AONB and the effects are not significant. In our view the landscape strategy should include measures to minimise and mitigate for potential impacts on RSPB Minsmere.

Landscape strategy

3.801 As we noted in our comments on the biodiversity net gain strategy we consider the Application should be focusing on landscape scale net gains and providing a functional ecological mosaic/ecotone and improving the actual ecological functionality of the surrounding area. We recommend the landscape strategy has a coordinated landscape scale approach linked into the Nature Recovery Network[403] and commitments related to this.

3.802 We recommend bigger, better and connected[404] landscape scale habitat creation to connect and extend habitats to benefit protected species such as

- Acid grassland, bare ground, scrub and ponds for turtle dove
- Acid grassland in closed undisturbed areas with no public access, no lighting, noise or cats for stone curlew
- Mixed species hedgerows for bats, turtle dove, reptiles
- Broadleaved woodland, with good understory and scrub margins for bats, especially barbastelle
- Areas of 'hands off' or low intensity management buffering key habitat types
- Wet woodland for scarce invertebrate assemblages
- Rough grassland for barn owl
- Nesting habitat for little tern and common tern
- Creation of lowland heathland, dry acid grassland and scrub mosaic to link up existing habitats across Priority Landscape to benefit multiple priority species including stone curlew, woodlark, nightjar, natterjack toad and turtle dove
- Creation and maintenance of lowland wet grassland to benefit breeding waders and wintering wildfowl

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400 RSPB Relevant Representation (RR-1059)

401 Suffolk Wildlife Trust Relevant Representation (RR-1180)

402 ES Vol. 2 Main Development Site Ch. 13 Landscape and Visual (APP-216)

403 https://www.gov.uk/government/publications/nature-recovery-network/nature-recovery-network

404 Lawton, J.H., Brotherton, P.N.M., Brown, V.K., Elphick, C., Fitter, A.H., Forshaw, J., Haddow, R.W., Hilborne, S., Leafe, R.N., Mace, G.M., Southgate, M.P., Sutherland, W.J., Tew, T.E., Varley, J., & Wynne, G.R. (2010) Making Space for Nature: a review of England's wildlife sites and ecological network. Report to Defra.

- Creation of reedbed and fen for breeding birds and scarce plant and invertebrate communities
- Adaption of coast to create more resilient, naturally functioning landscapes including inter-tidal mudflats, shingle and sand dunes, saltmarsh and coastal lagoons
- Improvement of wetland areas to provide improvements for wildlife and to provide natural flood management
- Creation of lineal connectivity between Sizewell Marshes SSSI and Minsmere

Habitat enhancement, reinstatement and creation

3.803 We are concerned the framework outlined in the Outline Landscape and Ecology Management Plan (oLEMP)[405] currently lacks sufficient details of baseline information, ecological objectives for habitats, species and ecological connectivity, habitat creation and management, robust monitoring and further interventions to be implemented if required and legal means of securing this throughout the lifetime of the development.

3.804 The oLEMP[406] paragraph 1.1.3 notes the oLEMP provides the framework for the Landscape and Ecological Management Plan which will be secured by requirement

> *The oLEMP provides the framework for the Landscape and Ecological Management Plan (LEMP) which will provide further details of the management measures and implementation of the habitat created, along with ongoing monitoring arrangements. This is secured by a requirement within Schedule 2 of the Draft DCO.*

3.805 Table 7.1 of the oLEMP[407] considers monitoring proposals for target communities before and after habitat enhancement, reinstatement or creation that will be included in the detailed LEMP and notes

> *Before and after enhancement, reinstatement or creation a full botanical species list and quality assessment should be carried out to monitor the success of restoration and as a baseline for monitoring, this should include the presence and abundance of species. The NVC may be an appropriate method for collecting data for monitoring or this may be bespoken to the target communities.*
>
> *…*
>
> *Target communities would be set for each habitat type for years 1, 2, 5 and 10.*

3.806 We consider target communities and monitoring methods should be defined and secured in the DCO.

3.807 Table 7.1 goes onto consider monitoring proposals for each habitat type, and for example for dry Sandlings grassland notes

> *Targets would be set for acid grassland according to the species list gathered before construction and thresholds identified for Section 41/Biodiversity Action Plan quality lowland meadow in the Countryside Stewardship Higher Tier Scheme made specific to the site, as well as the Joint Nature Conservation Committee guidance. Grassland habitat indicators would typically include the presence and abundance of key herb species present. This would weight desirable species against the injurious ones.*

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405 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)
406 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)
407 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)

3.808 The monitoring proposals lack detail and we consider detailed targets should be set and secured in the DCO.

3.809 The Applicant proposes creation of dry Sandlings grassland[408] from arable however this land will be under the construction site footprint for 10-12 years. We query how the Applicant envisages 10-12-year construction land use impacts will affect habitat establishment post construction.

3.810 We query the reason for semi-improved grassland[409] and propose scrub would be better if the soil type is not suitable for acid grassland.

Two Village Bypass

3.811 We query whether there may be opportunities for enhancements to provide habitat for protected species of bats and birds. We recommend an innovatively designed green bridge linking Foxburrow Wood County Wildlife Site with the woodland to the west. We also recommend additional tree planting to strengthen wildlife corridors.

Sizewell Link Road

3.812 The Sizewell Link Road remains a concern with regard the impact on barbastelle bats. Additional short lengths of hedgerow and approximately 0.17ha of additional woodland would be permanently lost as a result of the proposed changes[410]. This additional habitat loss further reduces habitat connectivity for bats and birds. We recommend additional hedgerow and tree planting to strengthen wildlife corridors.

Environmental Trust

3.813 Paragraph 1.1.6 of the Consultation Document for the Proposed Changes[411] proposed the establishment of an independent environmental trust to oversee the re-wilding and biodiversity of the Sizewell estate. We would need to understand this proposal in more detail. Plans for governance, implementation and ongoing financing will be required and we note that in order to be of most benefit to Suffolk's wildlife, plans for the estate should be targeted at declining locally-relevant habitats and species and should aim to increase connectivity with the wider landscape. We consider that this may sit better under the Applicant's corporate environmental and community responsibilities and other instruments would be more appropriate to address the impacts of the Sizewell C project. There is a need to ensure that legal requirements regarding protected sites and species are not compromised and that this aspiration is providing benefits over and above statutory requirements. We also request clarification of how this links in with the approach to biodiversity net gain.

Conclusions

3.814 We are concerned the Application will have a significant visual impact on visitor experience and the views available at RSPB Minsmere and the landscape strategy lacks measures to minimise and mitigate for this potential impact. In our view the landscape strategy should include measures to minimise and mitigate for potential impacts on RSPB Minsmere.

3.815 We consider the Application should be focusing on landscape scale net gains and providing a functional ecological mosaic/ecotone and improving the actual ecological functionality of the

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[408] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004) 4.1.26 – 4.1.32

[409] 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010) 5.1.6

[410] Volume 1 Environmental Statement Addendum Chapter 6 Sizewell Link Road (AS-185) 6.6.18

[411] Consultation Document Consultation on Proposed Changes

surrounding area. We recommend the landscape strategy has a coordinated landscape scale approach linked into the Nature Recovery Network and commitments related to this. We recommend bigger, better and connected landscape scale habitat creation to connect and extend habitats to benefit protected species.

3.816 We are concerned the landscape strategy lacks sufficient details of baseline information, ecological objectives for habitats, species and ecological connectivity, habitat creation and management, robust monitoring and further interventions to be implemented if required and legal means of securing this throughout the lifetime of the development.

4. LEGISLATION AND ASSOCIATED POLICY AND GUIDANCE

4.1 Please note that although the domestic legislation has now amended due to the UK leaving the EU, the caselaw and most of the policy and guidance has not. Therefore many of the quotes (particularly in caselaw) will have *old language*. We have left these as they are but tried to ensure new words, phrases etc are flagged in footnotes to ensure consistency with the legislation.

Outline Initial Submissions

4.2 Please note we have requested a further opportunity to make legal submissions towards the end of the Examination, acknowledging the helpful further discussions and documentation that has been promised by the Applicant and wishing to take account of for example further drafts of the DCO, s.106, plans and schemes as well as further requirements and measures.

4.3 Therefore these are very much our initial legal concerns.

The Ramsar Convention

4.4 Due to the presence of a Ramsar site, it is mentioning the Ramsar Convention on Wetlands of International Importance especially as Waterfowl Habitat 1971[412], which is an intergovernmental treaty that provides the framework for the conservation, protection and wise use of wetlands and their resources. And the longstanding Government policy that Ramsar sites have the same level of protection as SACs and SPAs: see paragraph 118 of the National Planning Policy Framework[413].

The Birds Directive

4.5 Due to the SSSI bird features and the SPA and its designated species, as well as the general duties set out below (including to ensure compliance with the Birds Directive), it is important to briefly consider the Birds Directive.

4.6 The main reasons for the Directive 2009/147/EC on the conservation of wild birds (the Birds Directive[414]) was the perilous state wild bird populations were in when the Directive was first introduced and the great need for the restoration of those populations. This represented a "serious threat to the conservation of the natural environment", with the measures to be taken needing to apply to *"the various factors which may affect the numbers of birds, namely the repercussions of man's activities"*. The conservation of wild bird species was identified as necessary to meeting the Community's objectives regarding the improvement of living conditions and sustainable development, and aimed at the long-term protection and management of natural

412 Ref. 1.4
413 Ref 1.14
414 Ref. 1.5

resources as an "*integral part of the heritage"* of the peoples of Europe and "to prevent commercial interests from exerting a possible harmful pressure on exploitation levels"[415].

4.7 Hence the preambles/recitals to the Birds Directives so clearly including this great concern as well as the Directive's aims and objectives, in particular paragraphs 3 and 8[416].

4.8 Unfortunately we find ourselves in a similar perilous state with species and habitats declines[417].

4.9 Unlike most species' protection, the Birds Directive provides protection for all wild birds (Article 1). By Article 2, Member States *"shall take the requisite measures to maintain the population of the species referred to in Article 1 at a level which corresponds in particular to ecological, scientific and cultural requirements while taking account of economic and recreational requirements, or to adapt the population of these species to that level."*[418] Article 5 requires Member States to take "requisite measures" to establish a general system of protection for all species of birds referred to in Article 1 prohibiting (inter alia) deliberate killing or capture by any method, or deliberate disturbance particularly during the period of breeding and rearing in so far as disturbance would be significant having regard to the objectives of the Directive.

4.10 These, along with the substantive provisions which give effect to those Article 2 obligations – namely Articles 3 and 4 – are the main protection provisions.

4.11 Article 4 includes a specific requirement for Annex I and migratory species, namely the classification of their 'most suitable territories' as SPAs, but the overall requirement for *special conservation measures concerning their* habitat *in order to ensure their survival and reproduction in their area of distribution* should not be forgotten (as discussed further below and its relevance for the consideration of the use of the SPAs and SSSIs by species other than those for which the sites were designated and notified).

4.12 The Joint Nature Conservancy Council's[419] guidance to SPA designations[420] highlighting the requirement to choose the most important sites within the UK confirms this by stating:

> *"As noted in the recitals to the Birds Directive (especially (3), (7) and (8)) and is clear from the structure of the Directive itself the overriding purpose of it was to arrest the decline in bird populations in Europe by imposing generalised population maintenance duties on Member States (Article 2) and to identify and classify the 'most suitable territories' for Annex I and migratory birds, to deliver an ecologically coherent network of SPAs (Article 4(1) and 4(2)) "in order to maximise the contribution of the sites to achieving favourable conservation status at the national, biogeographical or European level": Commission Note on Setting Conservation Objectives for Natura 2000 Sites (23 December 2013). "*

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415 Recital 6, 5, 7 and 9

416 (8) The preservation, maintenance or restoration of a sufficient diversity and area of habitats is essential to the conservation of all species of birds. Certain species of birds should be the subject of special conservation measures concerning their habitats in order to ensure their survival and reproduction in their area of distribution. Such measures must also take account of migratory species and be coordinated with a view to setting up a coherent whole.

417 Hayhow et al (2019) The State of Nature 2019. The State of Nature partnership. https://nbn.org.uk/wp-content/uploads/2019/09/State-of-Nature-2019-UK-full-report.pdf

418 There has not, as yet, been clear guidance on the interpretation of Article 2 (See, for example, what was said by Carnwath L.J. at §38 of *Sustainable Shetland v Scottish Ministers* [2015] UKSC 4, and at §17 identifying some apparent inconsistencies in CJEU case law. There are no set 'levels' in the UK for the purposes of Article 2

419 JNCC is the public body that advises the UK Government and devolved administrations on UK-wide and international nature conservation, provide evidence, information and advice

420 The UK SPA network: its scope and content, Volume 1: Rationale for the selection of sites – with the start of section 4 setting out the site selection process, section 4.5 (pgs 11- 12)

4.13 Article 4 does not, however, displace the wider, more general obligations, contained in Articles 1-2 and 5-9, but rather makes specific provision for birds within Annex 1 and migratory birds: see §19 of the CJEU's judgment in *R v Secretary of State for the Environment* [1997] Env LR 55.

4.14 Those obligations are generally implemented through the protections contained in the Wildlife and Countryside Act 1981 (as amended), but are also given effect through the obligations on appropriate authorities, nature conservation bodies and, in relation to marine areas, competent authorities in regulations 9 and 10 of the Conservation of Habitats and Species Regulations 2017 (as amended)(as discussed further below).

4.15 Importantly, the preambles/recitals to the Birds Directive and the structure of the Directive itself, highlight its purpose - to arrest the decline in bird populations in Europe by imposing protection requirements generally as well as specifically for Annex I and migratory species including the creation and protection of SPAs. But the requirements of course do not stop there as explained below.

SPA tightly drawn boundaries

4.16 It is important to note that the UK has transposed and implemented Article 4 through the creation of Special Protection Areas (SPAs) **only**, and therefore its approach to SPAs is relevant.

4.17 It is therefore worth noting here that unlike other countries, the UK does not include any buffer zones in and around its SPAs (nor its SACs and Ramsar sites) and therefore everything within those sites is included for ecological reasons. It is also notable that the UK has one of the lowest amounts of area designated as SACs or SPAs compared to other member states[421].

4.18 This means that any loss will directly impact the reasons for which the site was designated which makes it difficult for decision makers to be able to conclude that there will be no *adverse effect* if even a small part of that designated site is to be lost to the development.

The Habitats Directive

4.19 **The Habitats Directive**[422] requires EU Member States to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of community interest, which are listed under Annex I, II, IV and/or V.

4.20 The Habitats Directive's aim is to promote the maintenance of biodiversity throughout the EU through the restoration and maintenance of natural habitats and of wild fauna and flora (listed in its Annex I and II). The Habitats Directive (and its implementing domestic regulations) must be interpreted and applied by reference to the Precautionary Principle, which reflects the high level of protection pursued by European policy on the environment and in the light of its broad objective –namely a high level of protection of the environment.

4.21 Unlike the Birds Directive the Habitats Directive protects specific habitats and species by way of the creation of protected sites for them (SACs) and further provisions in relation to both European Protected Species (e.g. Natterjack toad and the bat species present and therefore of relevance here) and additional protections for priority species (not of relevance here).

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[421] European Environment Agency Website, Natura 2000 barometer https://www.eea.europa.eu/data-andmaps/dashboards/natura-2000-barometer. This is an official EU barometer for SPAs and was last updated in February 2011 – table and then graph on page 2.

[422] Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (Habitats Directive) 1992 - Ref. 1.6

4.22 We refer you to Natural England's advice in relation to European Protected Species legislative requirements including licensing. However we do wish to stress the importance and need to consider all required consents, licences and permissions whether included within the DCO and Deemed Marine Licence or not. As discussed briefly below applications much not be broken up into separate parts thus losing the vital need to consider cumulatively and in combination all possible effects arising out of the many moving parts of the proposed development. Therefore it must be considered, when determining this Application, as to whether consents, licences and permissions also required are likely to be granted or not.

Uncertainty and the Precautionary Approach

4.23 The Precautionary Principle exists for situations where scientific data does not exist or is incomplete and therefore it is not possible to complete a full evaluation of the possible risks a plan, project or activity may cause to the environment, including possible danger to humans, animal or plant health, or to the environment in general. The European Commission's Precautionary Principle guidance[423] states that it should apply when a phenomenon, product or process may have a dangerous effect, identified by a scientific and objective evaluation, if this evaluation does not allow the risk to be determined with sufficient certainty. As such the degree of precaution applied to an evaluation, or assessment, can be seen to be directly proportional to the extent of scientific uncertainty inherent in that assessment. As the guidance goes on to recommend:

> "*The implementation of an approach based on the precautionary principle should start with a scientific evaluation, as complete as possible, and where possible, identifying at each stage the degree of scientific uncertainty***."**

The Habitats Regulations[424]

4.24 In relation to plans and projects *likely to have an effect* on a European site (SPA, SAC and/or Ramsar site[425]), the Habitats Regulations set out the sequence of steps to be taken by the competent authority when considering authorisation for that plan or project before deciding to authorise that project. These are set out in regulation 63, as follows:

- Step 1: consider whether the project is directly connected with or necessary to the management of the SPA. If not
- Step 2: consider, on a precautionary basis, whether the project is likely to have a significant effect on the SPA, either alone or in combination with other plans or projects.
- Step 3: make an appropriate assessment of the implications for the SPA in view of its conservation objectives. Regulation 48(2) empowers the competent authority to require an applicant to provide information for the purposes of the appropriate assessment. There is no requirement or ability at this stage to consider extraneous (non-conservation e.g. economics) matters in the appropriate assessment.
- Step 4: consider whether it can be ascertained that the project will not, alone or in combination with other plans or projects, adversely affect the integrity of the SPA, having regard to the manner in which it is proposed to be carried out, and any conditions or restrictions subject to which that authorisation might be given (the Integrity Test).

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423 https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52000DC0001&from=EN

424 Ref. 1.6

425 And due to long standing government policy – Ramsar sites too

- Step 5: In light of the conclusions of the assessment the competent authority shall agree to the project only after having ascertained that it will not adversely affect the integrity of the SPA, alone or in combination with other plans or projects.

- Step 6: If, despite not being possible to ascertain that there will not be an adverse effect on the integrity of the site, there are no possible alternative solutions to the proposed development and there are imperative reasons of overriding public interest (IROPI) (which, subject to regulation 64(2), may be of a social or economic nature), for it, consent can still be granted if…

- Step 7: any and all necessary compensation measures must be secured to ensure the overall coherence and protection of the National Sites Network[426]. (consideration of the management objectives for the National Sites Network (as set out below) should be part of these considerations).

SPA and SAC Conservation Objectives

4.25 Under the Habitats Regulations, a site's Conservation Objectives are intrinsic to the Integrity Test when considering whether to grant consent for a plan or project – see Habitats Regulations 63(1).

4.26 In order to understand the Conservation Objectives in the context of Regulation 63(1) it is important to remind oneself of the role of SPAs under the Birds Directive – as set out above. These protected sites are part of the Member State's requirement for special conservation measures in order to ensure that their contribution to national and international "conservation status" of the species[427] is maximised: as set out in the headline words at the start of all Conservation Objectives:

4.27 Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;[428]

4.28 The Conservation Objectives are to be an articulation of the contribution that it is appropriate for the SPA to make in an enduring way. It would be inconsistent with the purposes of the protection and the role of SPAs to have SPA Conservation Objectives (or the interpretation of them) aiming for lower populations.

4.29 The role of SAC Conservation Objectives are similar but slightly different to reflect the differences in SAC and SPA classification and designation.

Principles for Undertaking an Appropriate Assessment

4.30 Taking a more detailed look at first Steps 3 and 4, the English and Welsh Court of Appeal in *R (on the application of Mynydd v Gwynt Ltd) v Secretary of State for Business, Energy and Industrial Strategy* [2018] EWCA Civ 231 has recently set out the following principles for appropriate assessments under Regulation 63(1) referring to other important caselaw in this area:

> *"(1) The environmental protection mechanism in Article 6(3) is triggered where the plan or project is likely to have a significant effect on the site's conservation objectives: Landelijke: Vereniging tot Behoud van de Waddenzee v Staatsscretaris van Lanbouw (Case C-127/02) [2005] All ER (EC) 353 at [42] ("Waddenzee").*

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426 Formerly known as the Natura 2000 Network

427 Please see points below on the management objectives of the National Sites Network and the requirements for both SPAs and SACS to ensure that the species are maintained and/or restored across their natural range

428 The SPA generic Conservation Objectives http://publications.naturalengland.org.uk/file/4811128974868480

(2) In the light of the precautionary principle, a project is "likely to have a significant effect" so as to require an appropriate assessment if the risk cannot be excluded on the basis of objective information: Waddenzee at [44].

(3) As to the appropriate assessment, "appropriate" indicates no more than that the assessment should be appropriate to the task in hand, that task being to satisfy the responsible authority that the project will not adversely affect the integrity of the site concerned. It requires a high standard of investigation, but the issue ultimately rests on the judgement of the authority: R (Champion) v North Norfolk District Council [2015] UKSC 52; [2015] 1 WLR 3710, Lord Carnwath at [41] ("Champion").

(4) The question for the authority carrying out the assessment is: "What will happen to the site if this plan or project goes ahead; and is that consistent with maintaining or restoring the favourable conservation status of the habitat or species concerned?": Sweetman v An Bord Pleanàla (Case C-258/11); [2014] PTSR 1092, Advocate General at [50].

(5) Following assessment, the project in question may only be approved if the authority is convinced that it will not adversely affect the integrity of the site concerned. Where doubt remains, authorisation will have to be refused: Waddenzee at [56-57]."

(6) Absolute certainty is not required. If no certainty can be established, having exhausted all scientific means and sources it will be necessary to work with probabilities and estimates, which must be identified and reasoned: Waddenzee, Advocate General at [107] and [97], endorsed in Champion at [41] and by Sales LJ in Smyth v Secretary of State for Communities and Local Government [2015] EWCA Civ 174 at [78] ("Smyth").

(7) The decision-maker must consider secured mitigation and evidence about its effectiveness: Commission v Germany (Case C-142/16) at [38].

(8) It would require some cogent explanation if the decision-maker had chosen not to give considerable weight to the views of the appropriate nature conservation body: R (Hart District Council) v Secretary of State for Communities and Local Government [2008] EWHC 1204 (Admin) at [49].

(9) The relevant standard of review by the court is the Wednesbury rationality standard, and not a more intensive standard of review: Smyth at [80]."

4.31 It is worth adding to this helpful review that an appropriate assessment requires all aspects of the project which could affect the site, its species and it's conservation objectives must be identified in the light of the best scientific knowledge in the field.[429] The competent authority, "taking account of the conclusions of the appropriate assessment of the implications…for the site concerned, in the light of the conservation objectives, are to authorise such activity *only if they have made certain* that it will not adversely affect the integrity of the site. That is the case where *no reasonable scientific doubt remains as to the absence of such effects*[430].

Site Integrity

4.32 The approach to the integrity question and its relationship to the SPA (and SAC) Conservation Objectives was explained by in *R (RSPB) v Defra and BAe* [2015] EWCA Civ 227 [2015] Env LR 24. It referred to the ECJ case of *Sweetman*[431] [5] which established that in judging integrity:

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[429] *Waddenzee* CJEU Case-127/02; [2004] ECR-7405 at [61].

[430] *Waddenzee* [59].

[431] *Sweetman v An Bord Pleanàla (Case C-258/11); [2014] PTSR 1092*

"it is the essential unity of site that is relevant. To put it another way, the notion of "integrity" must be understood as referring to the continued wholeness and soundness of the constitutive characteristics of the site concerned.

The integrity that is to be preserved must be that "of the site". In the context of a natural habitat site which has been designated having regard to the need to maintain the habitat in question at (or to restore it to) a favourable conservation status….

It follows that the constitutive characteristics of the site that will be relevant are those in respect of which the site was designated and their associated conservation objectives. Thus, in determining the whether the integrity of the site is affected the essential question the decision-maker must ask is "why was this particular site designated and what are its conservation objectives?"

4.33 Therefore, the removal or disturbance of any part of the site and its supporting habitat will make the populations for which the sites are designated more vulnerable and less able to recover.

4.34 In addition, it is important to add that the integrity of an SPA requires the lasting preservation of the constitutive characteristics of the site that are connected to the presence of the species whose preservation was the purpose of the designation of the SPA: *Sweetman* [39].

Functionally linked land

4.35 We welcome the Applicant's consideration of functionally linked land for some species, consistent with the NE Guidance and previous decisions regarding functionally linked populations (Chapman and Tyldesley, 2016[432]). (see for example Noise and Visual Disturbance section above).

4.36 However this consideration is not consistently applied for all SPA species. As the NE Guidance clearly explains[433],

"Twelve SPA cases related to terrestrial or coastal habitats involving a range of waterfowl. Seven related to the marine environment and sea birds. In all cases the decision maker recognised the potential importance of functionally linked land or sea and that it should be treated as part of the Habitats Regulations Assessment."

4.37 This represents a serious gap in both the assessment and consideration of possible effects on the SPA species including whether mitigation measures are required and possible.

Mitigation Measures

4.38 Whilst we appreciate the further discussions with the Applicant offered, we did wish to briefly cover there our concerns with the mitigation proposals currently before the Examination, in particular the need for any mitigation measures proposed (*due to possible adverse effects on the integrity of an European Site*), to be clear and certain in three ways:

- ecologically effective;
- financially secured; and
- legally secured.

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[432] Chapman, C. & Tyldesley, D. (2016) Functional linkage: How areas that are functionally linked to European sites have been considered when they may be affected by plans and projects - a review of authoritative decisions. Natural England Commissioned Reports, Number 207,

[433] see in particular electronic page 9 (unfortunately there are no page numbers in this part of the report and review of previous decisions report pages 29 -40

4.39 And it is for the Applicant to show that the mitigation it proposes will pass these certainty requirements.

4.40 The need for ecological certainty was recognised in *R (Champion) v North Norfolk District Council and another Champion [2015] UKSC 52,* paragraph 60, where the effectiveness of the mitigation measures proposed allowed the Supreme Court to determine, despite a procedural defect, that the decision should not be quashed. The information already available for the mitigation measures being sufficient and the lack of any concerns against their effectiveness is clear in the following paragraph:

> *"60 This was not a case where the environmental issues were of particular complexity or novelty. There was only one issue of substance: how to achieve adequate hydrological separation between the activities on the site and the river. It is a striking feature of the process that each of the statutory agencies involved was at pains to form its own view of the effectiveness of the proposed measures, and that final agreement was only achieved after a number of revisions. It is also clear from the final report that the public were fully involved in the process and their views were taken into account. It is notable also that Mr Champion himself, having been given the opportunity to raise any specific points of concern not covered by Natural England before the final decision, was unable to do so. That remains the case. That is not to put the burden of proof on to him, but rather to highlight the absence of anything of substance to set against the mass of material going the other way."*

Habitats Regulations General Duties

4.41 We would like to highlight, in particular, the obligation in regulation 9(3):

> *9.— Duties relating to compliance with the Directives*
>
> *(1) The appropriate authority, the nature conservation bodies and, in relation to the marine area, a competent authority must exercise their functions which are relevant to nature conservation, including marine conservation, so as to secure compliance with the requirements of the Directives.*
> *…*
> *(3) Without prejudice to the preceding provisions, a competent authority, in exercising any of its functions, must have regard to the requirements of the [Birds and Habitats] Directives so far as they may be affected by the exercise of those functions*.[434]

4.42 But the further duties in Regulation 10 are also important

> *10.— Duties in relation to wild bird habitat*
>
> *(1) Without prejudice to regulation 9(1), the appropriate authority, the nature conservation bodies and, in relation to the marine area, a competent authority must take such steps in the exercise of their functions as they consider appropriate to secure the objective in paragraph (3), so far as lies within their powers.*
> *....*
> *(3) The objective is the preservation, maintenance and re-establishment of a sufficient diversity and area of habitat for wild birds in the United Kingdom including by means of the upkeep, management and creation of such habitat, as appropriate), having regard to the*

[434] The terms of regulation 9(3) are not amended by the Conservation of Habitats and Species (Amendment) (EU Exit) Regulations although it needs to be read with the amended definitions of the relevant Directives and with the new regulation 9(4A) – regard must be had to any Secretary of State guidance – currently we do not believe this has been fully produced

requirements of Article 2 of the new Birds Directive (measures to maintain the population of bird species).

...

(7) In considering which measures may be appropriate for the purpose of securing or contributing to the objective in paragraph (3), appropriate account must be taken of economic and recreational requirements.

.....

(8) *So far as lies within its powers, a competent authority in exercising any function in or in relation to the United Kingdom must use all reasonable endeavours to avoid any pollution or deterioration of habitats of wild birds"*[435]

4.43 As mentioned above following the UK's departure from the EU these regulations have been changed and it is important to consider in particular the additions to them concerning the Management Objectives of the newly named National Sites Network[436]. Although these requirements already existed, it is helpful to have them clearly within our domestic legislation.

4.44 In summary regulation 16A, in Part 2 of the Regulations - **Conservation of Natural Habitats and Habitats of Species**, set out the requirements for the Network jointly and separately recognising the differences between SPAs and SACs (as set out above):

4.45 Authorities with relevant responsibilities must manage the National Site Network with a view to contributing to the achievement of the management objectives of it, namely

4.46 **For SACS** to maintain, or restore to favourable conservation status in their natural range

(i) the natural habitat types listed in Annex I to the Habitats Directive;
(ii) the species listed in Annex II to that Directive whose natural range includes any part of the United Kingdom's territory;

4.47 **For SPAs** to contribute, in their area of distribution, to ensuring the survival and reproduction of—

(i) the species of birds listed in Annex I to the new Wild Birds Directive;
(ii) regularly occurring migratory species of birds; and
(iii) to contribute, to securing compliance with regulation 9(1) (as set out above)

4.48 **For Both** take account of

(a) the importance of SACs and SPAs
(b) the importance of the sites for the coherence of National Site Network;
(c) the threats of degradation or destruction (including deterioration and disturbance of protected features) to which the sites are exposed; and
(d) in the case of migratory bird species, the importance of their breeding, moulting and wintering areas and staging points along their migration routes.

The Wildlife and Countryside Act 1981[437]

4.49 The Wildlife and Countryside Act 1981 (as amended) consolidates and amends existing national legislation and in part, implements the Bern Convention 1979 (Ref. 1.7), the Bonn Convention 1979 (Ref. 1.8) and the Birds Directive (Ref. 1.5). It creates the protection requirements for many species, as well as the ability to derogate from those requirements in certain limited

[435] Again the terms of regulation 10 are not amended by the Conservation of Habitats and Species (Amendment) (EU Exit) Regulations although it needs to be read with the amended definitions of the relevant Directives
[436] Formally known as the Natura 2000 Network
[437] Ref. 1.9

circumstance, helps controls the release of non-native species, and includes the need for and protection of Sites of Special Scientific Interest (SSSIs), among other things.

Sites of Special Scientific Interest

4.50 Sites of Special Scientific Interest (**SSSIs**) are the most important sites for national wildlife and natural features in England. In England, SSSIs are notified by Natural England who has a duty under Section 28 of the Wildlife and Countryside Act 1981 (as amended) (**the WCA**) to notify SSSIs where it is of the opinion that an area of land is of special interest by reason of any of its flora, fauna, or geological or physiographical features and to secure their day-to-day protection and conservation.

4.51 The purpose of SSSIs is defined in the Defra Code of Guidance[438] (paragraph 1) as being:

> *"…to safeguard, for present and future generations, the diversity and geographic range of habitats, species, and geological and physiographical features, including the full range of natural and semi-natural phenomena throughout England...".*

4.52 SSSIs make a fundamental contribution to the ecological processes upon which we all depend and to human quality of life. Individual SSSIs may also provide, or safeguard for the future, valuable research, educational and amenity resources.

4.53 Under Section 28G(2) of the WCA, public bodies must:

> *"…take reasonable steps, consistent with the proper exercise of the authority's functions, to further the conservation and enhancement of the flora, fauna or geological or physiographical features by reason of which the site is of special scientific interest".*

4.54 The Defra Code of Guidance (paragraph 73) states that the Secretary of State expects that all public bodies will take full account of their responsibilities under this duty whenever their actions may affect SSSIs.

4.55 Government Circular 06/2005[439], paragraph 61 requires all section 28G authorities, including local planning authorities, to apply strict tests when carrying out any functions within or affecting a SSSI, to ensure that they avoid or at least minimise adverse affects. It also requires public bodies to adopt the highest standards of management in relation to SSSIs in their ownership and to take positive steps wherever possible to conserve and enhance the special interest features of a SSSI where their activities may be affecting it or as opportunities arise in the exercise of their functions.

Energy Policy Background

4.56 The suite of Energy NPSs set out the Government's approach to ensuring the security of energy supplies and the policy framework within which new energy infrastructure proposals are to be considered. The presumption in favour of granting consent, as identified in NPS EN-1, *Overarching National Policy Statement for Energy*, is subject to the tests set out below in section 104 of the Planning Act (see NPS EN-1 paragraphs 4.1.2 and 1.1.2). Please see the SSSI loss section, above for consideration of the requirements in the Overarching National Policy Statement for Energy

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438 Defra (2003): Sites of Special Scientific Interest: Encouraging Positive Partnerships.

439 Government Circular 06/2005: Biodiversity and geological conservation – statutory obligations and their impact within the planning system, 16 August 2005. https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/7692/147570.pdf.

(EN-1)[440] and the National Policy Statement for Nuclear Power Generation (EN-6)[441] in respect of proposed land take from Sizewell Marshes SSSI.

4.57 **In addition the NPPF sets** out in a number of policies about the importance of local plan making and decision making in delivering sustainable development and meeting objectively assessed needs:

4.58 Paragraph 12 of the NPPF states:

> *"This National Planning Policy Framework does not change the statutory status of the development plan as the starting point for decision making. Proposed development that accords with an up-to-date Local Plan should be approved, and proposed development that conflicts should be refused unless other material considerations indicate otherwise. It is highly desirable that local planning authorities should have an up-to-date plan in place."*

4.59 Paragraph 14 of the NPPF states:

> *"At the heart of the National Planning Policy Framework is a presumption in favour of sustainable development, which should be seen as a golden thread running through both plan-making and decision-taking.*

Adding

> *For example, those policies relating to sites protected under the Birds and Habitats Directives (see paragraph 119) and/or designated as Sites of Special Scientific Interest; land designated as Green Belt, Local Green Space, an Area of Outstanding Natural Beauty, Heritage Coast or within a National Park (or the Broads Authority); designated heritage assets; and locations at risk of flooding or coastal erosion.*

4.60 In addition the NPPF includes conserving and enhancing the natural environment policies as set out in paragraph 117 as follows:

> *"To minimise impacts on biodiversity and geodiversity, planning policies should:*
>
> - *plan for biodiversity at a landscape-scale across local authority boundaries;*
> - *identify and map components of the local ecological networks, including the hierarchy of international, national and locally designated sites of importance for biodiversity, wildlife corridors and stepping stones that connect them and areas identified by local partnerships for habitat restoration or creation;*
> - *promote the preservation, restoration and re-creation of priority habitat, ecological networks and the protection and recovery of priority species populations, linked to national and local targets, and identify suitable indicators for monitoring biodiversity in the plan;*
> - *aim to prevent harm to geological conservation interests; and*
> - *where Nature Improvement Areas are identified in Local Plans, consider specifying the types of development that may be appropriate in these Areas."*

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[440] Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)

[441] Department of Energy and Climate Change (2011) National Policy Statement for Nuclear Power Generation (EN-6) Volume II of II - Annexes

4.61 And in relation to the presumption in favour of sustainable development (paragraph 14), paragraph 119 makes it clear that this presumption does not apply where development requiring appropriate assessment under the Birds or Habitats Directives is being considered, planned or determined.

The Biodiversity Duty

4.62 The Examining Authority needs to report on the duty under Section 40(1) of the Natural Environment and Rural Communities Act[442] (2006) for public authorities to have regard to conserving biodiversity. Section 40 states:

> *"Every public authority must, in exercising its functions, have regard, so far as is consistent with the proper exercise of those functions, to the purpose of conserving biodiversity."*

The EIA Requirements

4.63 We think it is important to consider the legislative requirements including the EIA Directive and its aims and objectives starting with the Directive's preamble/recitals, since the provisions of the our domestic legislation must, as far as possible, be interpreted in such a way that the purposes of the Directive are achieved.

Preamble

4.64 The Preamble highlights the need for:

> *"development consent for public and private projects which are likely to have significant effects on the environment should be granted only after prior assessment of the likely significant environmental effects of these projects has been carried out" and that "assessment must be conducted on the basis of the appropriate information supplied by the developer, which may be supplemented by the authorities and by the people who may be concerned by the project in question;"*

4.65 As well as that:

> *"the effects of a project on the environment must be assessed in order to take account of concerns to protect human health, to contribute by means of a better environment to the quality of life, to ensure maintenance of the diversity of species and to maintain the reproductive capacity of the ecosystem as a basic resource for life" (emphasis added).*

4.66 The importance of the public being informed and able to comment is key as helpfully emphasised by Lord Hoffmann in *Berkeley v SOSE* [2001] 2 AC 603 at 615c-616c[443].

The Infrastructure Planning (Environmental Impact Assessment) Regulations 2017[444]

4.67 Whilst we appreciate the Examining Authority is very familiar with these EIA requirements, it is worth highlighting the following, starting with Regulation 4, of the EIA Regulations and the need to consider environmental information:

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442 Natural Environment and Rural Communities Act (2006) https://www.legislation.gov.uk/ukpga/2006/16/section/40

443 We of course acknowledge that Lord Hoffmann's further remarks as to exercise of discretion now have to be read in the light of the comments of the Supreme Court in *Walton v Scottish Ministers* 2013 SC (UKSC) 67. However, the reference to *Berkeley* here is in relation to the need to gather all relevant environmental information including following public engagement, as set out above, a key requirement for EIAs.

444 https://www.legislation.gov.uk/uksi/2017/572/contents

"Prohibition on granting planning permission without consideration of environmental information

4(2) Where this regulation applies, the Secretary of State or relevant authority (as the case may be) must not (in the case of the Secretary of State) make an order granting development consent or (in the case of the relevant authority) grant subsequent consent unless an EIA has been carried out in respect of that application.

4.68 And it is important to note that Regulation 2 defines environmental information as:

""environmental information" means the environmental statement (or in the case of a subsequent application, the updated environmental statement), including any further information and any other information, any representations made by any body required by these Regulations to be invited to make representations and any representations duly made by any other person about the environmental effects of the development and of any associated development;"

4.69 With an "EIA" having the meaning given by regulation 5 – namely ;

(1) The environmental impact assessment ("the EIA") is a process consisting of—

(a) the preparation of an environmental statement or updated environmental statement, as appropriate, by the applicant;

(b) the carrying out of any consultation, publication and notification as required under these Regulations or, as necessary, any other enactment in respect of EIA development; and

(c) the steps that are required to be undertaken by the Secretary of State under regulation 21 or by the relevant authority under regulation 25, as appropriate.

4.70 With an "environmental statement" having the meaning given by regulation 14;

(2) An environmental statement is a statement which includes at least—

(a) a description of the proposed development comprising information on the site, design, size and other relevant features of the development;

(b) a description of the likely significant effects of the proposed development on the environment;

(c) a description of any features of the proposed development, or measures envisaged in order to avoid, prevent or reduce and, if possible, offset likely significant adverse effects on the environment;

(d) a description of the reasonable alternatives studied by the applicant, which are relevant to the proposed development and its specific characteristics, and an indication of the main reasons for the option chosen, taking into account the effects of the development on the environment;

…and

(f) any additional information specified in Schedule 4 relevant to the specific characteristics of the particular development or type of development and to the environmental features likely to be significantly affected.

(3) The environmental statement referred to in paragraph (1) must—

….(b) include the information reasonably required for reaching a reasoned conclusion on the significant effects of the development on the environment, taking into account current knowledge and methods of assessment;…

4.71 **With Schedule 4 of the EIA Regulations** setting out in more detail the "required information for inclusion in Environmental Statements" including

> ***3.*** *A description of the relevant aspects of the current state of the environment (baseline scenario) and an outline of the likely evolution thereof without implementation of the development as far as natural changes from the baseline scenario can be assessed with reasonable effort on the basis of the availability of environmental information and scientific knowledge.*
>
> ***5.*** *A* ***description*** *of the likely significant effects of the development on the environment resulting from, inter alia—*
>
> *(a) the construction* ***and*** *existence of the development, including, where relevant, demolition works;*
>
> *(e)* ***the cumulation of effects*** *with other existing and/or approved projects,* ***taking into account any existing environmental problems relating to areas of particular environmental importance*** *likely to be affected or the use of natural resources;*
>
> *…*
>
> *The description of the likely significant effects on the factors specified in regulation 5(2) should cover the direct effects and any* ***indirect, secondary, cumulative, transboundary, short-term, medium-term and long-term, permanent and temporary, positive and negative effects of the development****. This description should take into account the environmental protection objectives established at Union level (as they had effect immediately before exit day) or United Kingdom level which are relevant to the project, including in particular those established under the law of any part of the United Kingdom that implemented Council Directive 92/43/EEC and Directive 2009/147/EC.*
>
> ***6.*** *A* ***description of the forecasting methods or evidence,*** *used to identify and assess the significant effects on the environment,* ***including details of difficulties*** *(for example technical deficiencies or lack of knowledge) encountered compiling the required information and the* ***main uncertainties*** *involved.*
>
> ***7.*** *A description of the* ***measures envisaged to avoid, prevent, reduce or, if possible, offset*** *any identified significant adverse effects on the environment and,* ***where appropriate, of any proposed monitoring arrangements*** *(for example the preparation of a post-project analysis). That description* ***should explain the extent, to which*** *significant adverse* ***effects*** *on the environment* ***are avoided, prevented, reduced or offset****, and should cover both the construction and operational phases*. (emphasis added)

The Worst Case Scenario

4.72 As caselaw[445] has made clear the challenge for the EIA will be to ensure that all realistic and worst case variations of the project have been properly considered and clearly set out in the ES and the likely significant impacts from that worst case variation have been adequately assessed.

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[445] The 'Rochdale Envelope' arises from two cases: *R. v Rochdale MBC ex parte Milne* (No. 1) and *R. v Rochdale MBC ex parte Tew* [1999] and *R. v Rochdale MBC ex parte Milne* (No. 2) [2000].

All Aspects of the Proposed Development

4.73 The caselaw is very clear that all aspects of a project need to be considered together to ensure compliance with the EIA Regulations especially if those *other* parts are fundamental to the ability to achieve the main development – as is the case here.

4.74 We wish to stress that any assurances that possible cumulative impacts will be assessed separately during the consideration of the separate decisions is not, save in exceptional circumstances, a sufficient justification for not considering as part of the main Application and would result in a breach of the EIA requirements.

4.75 In addition the need to consider all potential cumulative effects arising from the Application as well as the potential in-combination effects from other plans and projects. Currently both these appear to be lacking in the environmental statement as set out above in several of the sections.

4.76 As discussed in each section and below – we continue to have serious concerns about the level of data available, the assessment undertaken and mitigation measures proposed including their effectiveness.

4.77 We look forward to continuing discussions with the Applicant but at this stage of the Application process the lack of details and firm commitments is of great concern

The Development Consent Order - Initial Concerns

4.78 Currently we have two broad issues in relation to the all important DCO and Deemed Marine Licence[446]

(1) Schedule 2 requirements

4.79 Article 2 grants development consent subject to the provisions of the Order and of Schedule 2 requirements (appreciating that these are akin to planning conditions). If we have understood correctly, the requirements of most relevance and concern are Requirements 2 (Code of Construction Practice) 4 (Terrestrial ecology management plan) 5 (Surface and foul water drainage) and 7 (Water Management) but also 13 (Ancillary structures, other buildings and plan) and 14 (Landscape works).

4.80 Requirement 2 requires construction (etc) of the authorised development to be carried out in "general accordance" with the Code of Construction unless otherwise agreed by the local planning authority. We understand that the Code of Construction Practice[447] is the relevant Code. However we do question why just “general accordance” rather than in accordance with and where precisely is there flexibility in that Code?

4.81 Requirement 4 - this only refers to ecology monitoring. It appears from the Mitigation Route Map [448]) that other ecological mitigation measures are to be secured through the Code of Construction Practice (Requirement 2), through scheme design (Article 3) through landscape requirement (Requirement 14) (and others) or obligations in the Section 106. However we note that the Examining Authority are seeking information in this respect from the Applicant - see EX Q1 Part 2 Bio 1.9: We will review answers provided and comment further at that stage.

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446 https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010012/EN010012-001671-SZC_Bk3_3.1_Draft_Development_Consent_Order.pdf

447 ref 8.11

448 document 8.12, REP1-006

4.82 In addition there are many cases in every chapter of the ES, Volume 2, chapter 14 (for example) where it is stated that primary and tertiary mitigation with the aim of reducing or lowering levels of environmental effects (as set out in detail in the above sections). Inevitably the lists of primary and tertiary mitigation vary from site to site and receptor to receptor, however we remain extremely concerned as to how the Examining Authority can be sure that all the primary and tertiary mitigation listed is secured and will be delivered.

4.83 Therefore it is important that the Applicant clearly explains where and how the descriptions of such mitigation in the ES volumes and relevant chapters is reconciled with the mitigation secured in the DCO and the s.106 agreement

4.84 Currently, there is not, for example, a mitigation equivalent to the Terrestrial Ecology Monitoring Plan in requirement 4 nor a construction specific Ecology Management and Monitoring Plan.

4.85 We note PINS Advice Note 15is very clear in [449]this regard:

> *"16.1 An application may have significant adverse environmental effects that require mitigation; such effects will be identified in the accompanying ES and/ or relevant environmental information.* ***Any mitigation measures relied upon in the ES must be robustly secured and this will generally be achieved through Requirements in the draft DCO.*** *Mitigation that is identified in the ES as being required must also be clearly capable of being delivered."* (emphasis added)

4.86 It is not immediately apparent to us whether - or if so how - this has been done here

4.87 Requirement 7 does refer to the "the adjacent Sites of Special Scientific Interest". And in addition to the points made above (please see hydrology section) and all relevant information being required now, we also question due to the involvement of those adjacent SSSIs why Natural England is not also a required consultee. We would also request that due to the involvement of the RSPB and SWT, we too should be consulted.

4.88 We also note that there is no general requirement that the authorised development be carried out in accordance with the documents to be certified by the Secretary of State under Schedule 22 – this appears to be a fundamental oversight.

Deemed marine licence

4.89 We have two main questions at this stage

(1) What is the Marine Environmental Management Plan referred to in Schedule 20 Part 4 para 18? There appears to be no definition in Schedule 20, and

(2) When will this plan be produced to enable interested parties such as us to review and comment on it. As far as we are aware, there is not even an outline of what it is to cover, nor clarity on whether sufficient detail will be provided in the Costal Processes Monitoring and Mitigation Plan (as discussed in detail above) to check for shared issues.

[449] https://infrastructure.planninginspectorate.gov.uk/wp-content/uploads/2014/10/advice_note_15_version_1.pdf

The s.106 Agreement[450]

Schedule 9 - Implementation Plan

4.90 As far as we are aware a draft of the Implementation plan referred to within this Schedule has not been provided. This appears to be a key document including timetables proposed for secure avoidance/mitigation measures and would need to be consistent with other plans discussed above or clear how the various plans will work together as part of it

4.91 Also we note that the Applicant is to keep the Councils informed of progress (para 2.3). We would suggest that this is also relevant for the Environmental Review Group and Environmental Working Groups especially due to their role in keeping the implementation of Key Environmental Mitigation under review

4.92 We would also question whether it should be the Applicant alone who determines the appropriate course of action in the event of any delay Delays (as set out in paragraphs 2.4 - 2.8, particularly paragraph 2.6). We are particularly concerned about the potential for some delays to cause additional or further effects including due to delays in mitigation measure. As mentioned above our concerns with the adequacy of the assessments and the lack of detail currently on the proposed mitigation measures, adds to this concern.

4.93 We have further concerns namely:

4.94 Whether the drawing forward of Contingent Funds will be sufficient

4.95 What role is envisaged for Environmental Review Group and the Environmental Working Groups Why is it only the Planning Group (Schedule 17 para 5.1.2 - comprises 1 representative from each of the Councils and up to two representative from the Applicant) and Delivery Steering Group (Schedule 17 para 2 - comprises one service director (or equivalent) from each of the Councils and again up to 2 representatives from the Applicant) that are to be notified of any review and its outcomes

4.96 The make up of these groups is also concerning in relation to any changes required to the implementation of Key Environmental Mitigation as it appears this could be determined by the Applicant, appreciating the involvement of and notification to the Councils but there will be no working majority of non Applicant members.

Schedule 11 - Natural Environment

The Environment Review Group

4.97 Again we have concerns about how the different aspects covered by Schedule 11 (and Schedule 9 and 17) will work and the membership of Environmental Review Group is helpful with a working majority of non Applicant members, the issue may arise about when referred to Delivery Steering Group.

4.98 We do however query why the Environmental Review group will only meet every 6 months given that progress in carrying out Key Environmental Mitigation is to be reported to the Councils at least quarterly (Sch 11 para 2.3)

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450 https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010012/EN010012-003959-Sizewell%20C%20Project%20-%20If%20needed,%20draft%20section%20106%20Agreement(s)%20(s.106),%20s.106%20Explanatory%20Memorandum%20and%20draft%20Confirmation%20and%20Compliance%20Document.pdf

Ecology Working Group

4.99 A key part of its role appears to be to review monitoring undertaken with the various Monitoring Plans (defined in para 1 to the Schedule). What appears to be lacking is provision for any technical expert input or consultation.

Legal Conclusions

4.100 We will continue to work with the Applicant on these concerns and provide comments on further information submitted to the Examination for example further details on the required management mitigation, compensation and monitoring plans and schemes as well as the details of the various funds mentioned above. We will update the Examining Authority of our position in light of any new information presented

4.101 However for now (as also covered in other section of these written representations) we have serious concerns about the adequately of information currently available and strongly recommend that without those details including the lack of clarity on the required legal, financial and ecological securing of requirements, measures, plans and schemes – confidence in them is premature.

5. BIODIVERSITY NET GAIN

Summary

5.1 Our concerns relate to the Applicant's approach to biodiversity net gain (BNG).

5.2 Whilst we welcome the Applicant including BNG within their proposals, despite the introduction of BNG within the Environment Bill not yet being a specific requirement (due to the Bill having not passed nor the further work required to establish a BNG system) we do not agree the Application can achieve net gain due to direct adverse impact on Sizewell Marshes SSSI from loss of a significant proportion of the SSSI, nor the amount claimed by the Applicant.

5.3 We are grateful to the Applicant for the discussions held and agree that the Application should take account of the requirements the new net gain system will include once established.

5.4 In addition there is a Natural England requirement for NSIPs to contribute significantly to environmental net gain[451]. As such, the Application should be demonstrating a high biodiversity metric score that does not rely on protected species mitigation and compensation measures to get over the 10% uplift requirement.

5.5 The biodiversity metric 2.0[452] used for the assessment is a consultation draft that contains software bugs and grey areas. The Applicant is currently taking full advantage of all grey areas to increase the metric score including revised baselines, use of measures that are for protected sites and species mitigation and compensation, heavy use of 'fairly good' category whilst ignoring limitations and trading down. The choices made have in each case increased the claimed BNG.

5.6 Our main concerns include:

- use of measures that are for protected sites and species mitigation and compensation

[451] The Planning Inspectorate Advice Note 11, Annex C – Natural England and the Planning Inspectorate

[452] Crosher, I. A, Gold, S. B., Heaver M.D., Heydon M.A., Moore L.D., Panks, S.A., Scott S.C., Stone D.A. and White N.A. (2019) The Biodiversity Metric 2.0: auditing and accounting for biodiversity value. User guide (Beta Version, July 2019). Natural England

- the baseline policy is directly at odds with the biodiversity gain system to be introduced by the Environment Bill
- the time for habitats to reach target condition (temporal risk) does not take account of the construction phase
- the replacement of higher biodiversity habitats with those of lower biodiversity (trading down)
- lack of evidence to support the likelihood of delivering the predicted on-site habitat conditions on post-construction soils
- lack of evidence to support the likelihood of delivering the predicted off-site habitat conditions post development
- the % net change in the metric score is calculated versus the on-site baseline but it should be calculated versus the total on-site plus off-site baseline
- the biodiversity value of existing habitats has not been adequately considered and taken account of
- heavy use of 'fairly good' category

5.7 We consider the key finding from the Applicant's BNG proposals to be at best only just over the 10% boundary by a narrow margin, and that if any one of several marginal decisions were reversed it would fall below it. Therefore it is our view that the Application is not making the ecological contribution claimed nor that is proportional to the size of it - at best it is providing no more than the minimum standard expected of any housing development[453] [454].

5.8 We also consider it important to note the biodiversity metric is not suitable for tracking ecological delivery over time - it is designed to be a one-time snapshot of future commitments at the point of the planning application and does not work as a tracking tool (e.g. the score will rise over time even if condition targets are missed).

5.9 We are also concerned the Applicant does not appear to have identified a legal mechanism for securing a 10% metric score in their mitigation strategy.

Introduction

5.10 We have engaged with the Applicant throughout the pre-application and pre-examination period. We reviewed the now superseded Biodiversity Net Gain report[455] for the main development site (MDS) submitted with the DCO application and expressed our concerns in our relevant representations [RR-1059 (RSPB) and RR-1180 (SWT)].

5.11 We have reviewed the net gain reports for the main development site[456], two village bypass[457] and Sizewell C Link Road[458] submitted at Deadline 1 and are pleased that some of our concerns around methodology have been addressed. We outline our current position below.

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453 Ministry of Housing, Communities and Local Government (2019) National Planning Policy Framework (NPPF) paragraph 175

454 A Green Future: Our 25 Year Plan to Improve the Environment

455 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (APP-266)

456 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

457 9.6 Biodiversity Net Gain Report - Two Village Bypass (REP1-018)

458 9.5 Biodiversity Net Gain Report - Sizewell Link Road (REP-017)

Approach to biodiversity net gain and direct adverse impact on Sizewell Marshes SSSI

5.12 Whilst the RSPB and SWT are fully aware of how the Environment Bill[459], will introduce a requirement for BNG, currently that legislation is not passed and there is a good chance it will not have done so by the time this Examination closes. In addition, the Bill purely introduces a framework for a BNG system and much more is needed before that system can get up and running.

5.13 For these reasons we would suggest that reference to the Bill (for example paragraph 1.5.2 of the Biodiversity Net Gain Report[460]) is premature. However it is useful to consider how BNG is described within it and its helpful explanatory notes. For example, at paragraph 1575[461] of the Bill's explanatory notes broadly excludes any development from BNG if it has a direct adverse impact on a protected site

> *Whilst it is generally agreed in practice that development cannot claim biodiversity net gain in cases when development results in land take from statutory protected sites (such as Sites of Special Scientific Interest, Special Protection Areas, Special Areas of Conservation, and Ramsar sites), development on such sites is not specifically exempted from the net gain requirement.*

5.14 The Bill[462] at paragraph 1575 of its explanatory notes[463] also notes the biodiversity net gain requirement for development on such sites is additional to any existing legal or policy requirements for statutory protected areas and their features

> *The biodiversity metric does not address impacts on species, recognise the significance of site designations, take account of indirect impacts, cumulative impacts or in combination impacts. In recognition of these limitations, the biodiversity net gain requirement for development on such sites is additional to any existing legal or policy requirements for statutory protected areas and their features, including restoration and conservation of designated features and the achievement of favourable conservation status and favourable condition. These requirements will need to be dealt with separately by the developer and planning authority.*

5.15 The Overarching National Policy Statement for Energy (EN-1)[464] also notes at paragraph 5.3.11 where proposed development is likely to have an adverse effect on an SSSI development consent should not normally be granted

> *Where a proposed development on land within or outside an SSSI is likely to have an adverse effect on an SSSI (either individually or in combination with other developments), development consent should not normally be granted. Where an adverse effect, after mitigation, on the site's notified special interest features is likely, an exception should only be made where the benefits (including need) of the development at this site, clearly outweigh both the impacts that it is likely to have on the features of the site that make it of special scientific interest and any broader impacts on the national network of SSSIs. The* [PINs] *should use requirements and/or planning obligations to mitigate the harmful aspects of the development and, where possible, to ensure the conservation and enhancement of the site's biodiversity or geological interest.*

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459 The Environment Bill, May 2021 - https://bills.parliament.uk/publications/41447/documents/196

460 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

461 The Environment Bill Explanatory Notes, May 2021 - https://bills.parliament.uk/publications/41448/documents/197

462 The Environment Bill, May 2021 - https://bills.parliament.uk/publications/41447/documents/196

463 The Environment Bill Explanatory Notes, May 2021 - https://bills.parliament.uk/publications/41448/documents/197

464 Department of Energy and Climate Change (2011) Overarching National Policy Statement for Energy (EN-1)

5.16 As outlined in the SSSI loss section above we do not agree the Applicant has demonstrated compliance with all legal and policy requirements for justification of the principle and scale of the proposed loss of part of Sizewell Marshes SSSI against the policy tests set out in EN-1 (Overarching NPS for Energy).

5.17 Although the National Planning Policy Framework[465] (NPPF) (2019) does not contain specific policies for nationally significant infrastructure projects it does note at paragraph 170 that planning decisions should minimise impacts on and provide net gains for biodiversity

> *Planning policies and decisions should contribute to and enhance the natural and local environment by:*
>
> *….*
>
> *d) minimising impacts on and providing net gains for biodiversity, including by establishing coherent ecological networks that are more resilient to current and future pressures;*

and at paragraph 175 notes opportunities to secure measurable net gains for biodiversity should be encouraged

> *When determining planning applications, local planning authorities should apply the following principles:*
>
> *d) opportunities to incorporate biodiversity improvements in and around developments should be encouraged, especially where this can secure measurable net gains for biodiversity.*

5.18 BNG Good Practice Principles for Development[466] notes at section 9.5 that projects affecting statutory designated sites cannot achieve BNG

> *Projects negatively affecting statutory designated sites or irreplaceable habitats cannot as a project achieve BNG.*

5.19 We do not agree the Application can achieve net gain due to direct adverse impacts on Sizewell Marshes SSSI from loss of a significant proportion of the SSSI. Despite this, we agree the Application should be meeting the requirements of the net gain system and our comments below are in relation to that achievement.

Biodiversity net gain and landscape strategy

5.20 Annex C to the Planning Inspectorate Advice Note 11[467] notes

> *Natural England will seek opportunities for positive environmental outcomes from major infrastructure developments. NSIPs can make a significant contribution to delivering the environmental ambition in the Government's 25 Year Environment Plan (25YEP). This aims to deliver an environmental net gain through development and infrastructure.*

5.21 The Defra Biodiversity Offsetting Pilots guidance[468] at paragraph 17 notes

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465 Ministry of Housing, Communities and Local Government (2019) National Planning Policy Framework (NPPF)

466 Baker, J., Hoskin, R. and Butterworth, T. (2019) Biodiversity net gain. Good Practice Principles for Development. A practical guide.

467 Annex C to the Planning Inspectorate Advice Note 11: Working with public bodies in the infrastructure planning process – Natural England and the Planning Inspectorate

468 Defra (March 2012) Biodiversity Offsetting Pilots Technical Paper: the metric for the biodiversity offsetting pilot in England

Offsetting should:

- *contribute to enhancing England's ecological network by creating more, bigger, better and joined areas for biodiversity (as discussed in Making Space for Nature[469])*

5.22 BNG Good Practice Principles for Development[470] states at Technical Note T5 section T5.2 projects of national significance should focus on landscape scale net gains

Projects of national significance should underpin and support the delivery of UK and devolved government biodiversity commitments and should focus on landscape scale net gains and a contribution to national biodiversity targets.

5.23 Therefore the Application's BNG proposals should be focusing on landscape scale net gains and providing a functional ecological mosaic/ecotone and improving the actual ecological functionality of the surrounding area. We also query whether there may be opportunities for net gain habitat enhancements to provide habitat for protected species of bats, birds (barn owl[471], turtle dove, stone curlew, nesting terns), mammals and invertebrates.

5.24 Table 13 of the Biodiversity Net Gain Report[472] shows arable and amenity planting provide 106.8 biodiversity units, against 758.76 total for the main development site post development. It would be expected that habitat creation would focus on maximising biodiversity and we query the inclusion of significant areas of low ecological value habitats in the metric score. If arable land is to be created, we request inclusion in highest tier agri-environment scheme or a similar commitment to biodiversity enhancements such as field margins and hedgerow management.

Biodiversity net gain and protected species

5.25 We are concerned that potential loss of habitat for birds has not been adequately assessed by the Applicant particularly on the associated development sites (for example the two village bypass and link road as discussed below). We consider there should be sufficient bird surveys in accordance with best practice guidelines[473] to establish the current assemblage using the main development site and each associated development site to inform the assessment of potential losses and measures to retain and enhance the bird assemblage to ensure there is demonstrable net gain. Natural England also expressed concerns over the adequacy of breeding bird surveys[474].

5.26 We welcome any additional measures not required for mitigation and/or compensation that would be implemented to benefit specific species.

Inclusion of mitigation and compensation measures

5.27 As stated above, we do not agree the Application can achieve net gain due to direct adverse impacts on Sizewell Marshes SSSI. Despite this, we agree the Application should be meeting the requirements of the net gain system and should be providing landscape scale net gain proportionate to the scale of the project in line with planning policy. As such, the Application

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469 Lawton, J.H., Brotherton, P.N.M., Brown, V.K., Elphick, C., Fitter, A.H., Forshaw, J., Haddow, R.W., Hilborne, S., Leafe, R.N., Mace, G.M., Southgate, M.P., Sutherland, W.J., Tew, T.E., Varley, J., & Wynne, G.R. (2010) Making Space for Nature: a review of England's wildlife sites and ecological network. Report to Defra.

470 Baker, J., Hoskin, R. and Butterworth, T. (2019) Biodiversity net gain. Good Practice Principles for Development. A practical guide.

471 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology (AS-033) 14.12.40 to 14.12.42

472 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

473 Bird Survey & Assessment Steering Group. (2021) Bird Survey Guidelines for assessing ecological impacts, v.0.1.0. https://birdsurveyguidelines.org

474 Natural England Relevant Representation [RR-0878] 37., 113., 114., 115., 176. and 183.

should be demonstrating a high metric score that does not rely on protected species mitigation and/or compensation habitat to get over the 10% uplift requirement.

5.28 We are concerned that protected sites and species mitigation and compensation measures have been double counted in the metric score without first demonstrating the adequacy of those mitigation and compensation measures and then providing net benefits for biodiversity over and above requirements for avoidance, mitigation and/or compensation measures.

5.29 Paragraph 2.9.1 of the Biodiversity Net Gain Report[475] notes mitigation areas are included

> *Landscape design has been carried out to maximise the benefits to a range of species, maximise biodiversity value and provide ecosystem service benefits. The design is layered in this way, for example, the reptile mitigation areas also provide benefits to a range of other species, for example bats and invertebrates. BNG as presented in the biodiversity metric guidance is a holistic landscape scale assessment of the overall long-term status of the site and the habitats within. As such, it is appropriate that where mitigation for species which results in a change of habitat (such as within the Studio Fields Complex) is assessed within the biodiversity metric.*

5.30 We consider 'no net loss' activities such as mitigation and compensation should not be relied upon to achieve the 10% metric target. We consider there should be no double counting if a mitigation or compensation area has benefit to other species unless specific measures are implemented for those other species.

5.31 Section 5.1[476] notes the marsh harrier compensation area and reptile habitat areas are included in the off-site biodiversity calculations. The marsh harrier compensation area is incorrectly described as mitigation however it provides compensatory foraging habitat for marsh harrier. The reptile habitat areas provide mitigation habitat for reptiles. One of these, Great Mount Walk, is within the marsh harrier compensation area. Therefore the Applicant is triple counting this area as reptile mitigation, marsh harrier compensation and BNG. As stated above we consider there should be no double counting (or triple counting) if a mitigation or compensation area has benefit to other species unless specific measures are implemented for those other species.

5.32 CIEEM (2018) guidelines[477] describe the key principles of ecological impact assessment at section 1.19 as

Avoidance	*Seek options that avoid harm to ecological features (for example, by locating on an alternative site).*
Mitigation	*Negative effects should be avoided or minimised through mitigation measures, either through the design of the project or subsequent measures that can be guaranteed – for example, through a condition or planning obligation.*
Compensation	*Where there are significant residual negative ecological effects despite the mitigation proposed, these should be offset by appropriate compensatory measures.*

[475] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[476] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[477] CIEEM (2018) Guidelines for Ecological Impact Assessment in the UK and Ireland: Terrestrial, Freshwater, Coastal and Marine version 1.1. Chartered Institute of Ecology and Environmental Management, Winchester

Enhancement *Seek to provide net benefits for biodiversity over and above requirements for avoidance, mitigation or compensation.*

5.33 The Applicant should correctly apply the mitigation hierarchy:

1. Avoidance: demonstrate measures to avoid impacts on protected species and habitats
2. Mitigation: demonstrate reptile mitigation measures are adequate and those within Great Mount Walk are additional to those for marsh harrier compensation
3. Compensation: demonstrate the marsh harrier compensation measures are adequate and additional to those required for reptile mitigation
4. Net gain: provide additional enhancements to benefit other species (or ecological connectivity) over and above avoidance, mitigation and compensation to provide net gain

5.34 Paragraph 4.2 of the Marsh Harrier Mitigation Area Feasibility Report[478] describes the design objectives for creation of lowland heath at Great Mount Walk (within the marsh harrier compensation area)

- *Sow short sward acid grassland mix.*
- *Spread heather brashings of local (Suffolk) provenance in suitable locations to encourage heather establishment as well as plant and invertebrate species indicative of Suffolk Sandlings heath.*
- *Create short sward acid grassland in Great Mount Walks.*
- *Provide log and brash piles to encourage populations of small mammals in the short term as vegetation develops.*
- *Allow development of small areas (less than 0.5ha) of bare ground to encourage development of a diverse assemblage of arable weeds and invertebrate species.*
- *Creation of lowland heath is a long-term process and should begin as early as is practicable to enable the establishment of vegetation and development of a good habitat structure of benefit to nesting birds and small mammals.*
- *Management of scrub and heath will be required to maximise the populations of breeding farmland bird species that are present.*

5.35 These measures are required to provide compensatory foraging habitat for marsh harrier and cannot be counted in the metric score.

5.36 The Reptile Mitigation Strategy Figure 14C2A.17[479] shows hay piles, log/brush piles and hibernacula within Great Mount Walk but it is not clear which of these are provided as reptile mitigation over and above requirements for the marsh harrier foraging area. Reptile mitigation also cannot be counted in the metric score.

5.37 The Biodiversity Net Gain Report[480] provides no indication of specific measures for other species over and above protected species mitigation and compensation measures.

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478 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14C5 Marsh Harrier Mitigation Area Feasibility Report (APP-259)

479 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14C2A Reptile Mitigation Strategy Figures 14C2A.1 - 14C2A.18 (APP-255)

480 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

5.38 We consider 'no net loss' activities such as mitigation and compensation should not be relied upon to achieve the 10% metric target. We consider there should be no double counting (or triple counting) if a mitigation or compensation area has benefit to other species unless specific measures are implemented for those other species. We of course welcome any additional measures not specifically required for mitigation and/or compensation that would be implemented to benefit specific species without impeding the main purposes of the marsh harrier compensation area and the reptile mitigation areas, but their use within the metric target. Appendix B of the Biodiversity Net Gain Report [REP1-004] shows the offsite areas with some of the reptile mitigation areas annotated. We request that a map with all the mitigation and compensation areas clearly annotated is provided.

Biodiversity metric 2.0

5.39 We consider it important to note the metric is not suitable for tracking ecological delivery over time - it is designed to be a one-time snapshot of future commitments at the point of planning application and does not work as a tracking tool (e.g. the score will rise over time even if condition targets are missed).

5.40 The Outline Landscape and Ecology Management Plan (oLEMP)[481] Table 7.1 notes monitoring proposals refer to biodiversity net gain which suggests the Applicant may intend to use the metric to track ecological delivery over time

> *monitoring with regards to achieving the desired communities and quality as demonstrated in the biodiversity net gain report (ES Volume 2 Chapter 14 Appendix 14E).*

5.41 We request clarification that the Applicant does not intend to use the metric as a tracking tool.

Mechanism for securing net gain

5.42 We are concerned the Applicant does not appear to have identified a legal mechanism for securing a 10% metric score in their mitigation strategy.

5.43 The oLEMP[482] notes at paragraph 1.1.3 the Landscape and Ecological Management Plan will be secured by requirement

> *The oLEMP provides the framework for the Landscape and Ecological Management Plan (LEMP) which will provide further details of the management measures and implementation of the habitat created, along with ongoing monitoring arrangements. This is secured by a requirement within Schedule 2 of the Draft DCO.*

and at paragraph 4.1.7 notes

> *The Biodiversity Net Gain report …. demonstrates that net gain would be achieved if the habitats described within this oLEMP are delivered as proposed.*

and at Table 7.1 notes monitoring proposals refer to biodiversity net gain

> *monitoring with regards to achieving the desired communities and quality as demonstrated in the biodiversity net gain report (ES Volume 2 Chapter 14 Appendix 14E).*

[481] 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)
[482] 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)

5.44 Requirement 14 of the draft DCO[483] notes the landscape scheme for the permanent development site to be approved by East Suffolk Council must contain

> *(vii) a landscape and ecology management plan which will be prepared in general accordance with the measures set out in the Outline Landscape and Ecology Management Plan.*

5.45 We are however concerned that Schedule 2 of the Draft DCO[484] does not include a specific requirement for securing a 10% metric score in accordance with the draft Environment Bill [Schedule 14, 9(3) and 10(1) as currently drafted] and planning policy.

5.46 EN-1 (Overarching NPS for Energy) notes at paragraph 5.3.11 the [PINs] should use requirements and/or planning obligations to ensure enhancement of biodiversity

> *The* [PINs] *should use requirements and/or planning obligations to mitigate the harmful aspects of the development and, where possible, to ensure the conservation and enhancement of the site's biodiversity or geological interest.*

5.47 CIEEM (2018) guidelines[485] at paragraph 6.13 notes enhancements should be designed to deliver biodiversity objectives specified in relevant policy documents and evidence should be provided to support the likelihood of delivering the predicted benefit

> *Enhancement measures should be designed to deliver biodiversity objectives that are specified in relevant policy documents, and evidence should be provided to support the likelihood of delivering the predicted benefit. They should be incorporated into scheme design and assessed within the EcIA. To ensure that enhancements are enduring, their delivery and management should normally be guaranteed through a legal obligation, such as, in England and Wales, a planning obligation under section 106 of the Town and Country Planning Act 1990116 , or its equivalent provision elsewhere.*

5.48 Paragraph 7.12 notes

> *All parties should understand the actions they need to take during the implementation stages of a project. This will mean identifying and designing in detail the measures necessary to avoid, mitigate and compensate negative effects, and any measures necessary to achieve enhancements. Details of these measures will be set out in the EcIA and may be presented in an Environmental Management Plan (EMP) or similar document.*

5.49 We request the Applicant provides full details of all the measures necessary for protected species mitigation and compensation and full details of additional measures to achieve enhancements for example to benefit other species. We also request evidence is provided to support the likelihood of delivering the predicted benefits. The enhancement measures must be legally secured and we consider Schedule 2 of the Draft DCO[486] should include a specific requirement for enhancement measures to secure a 10% metric score in accordance with the draft Environment Bill (Schedule 14, 9(3) and 10(1) as currently drafted) and planning policy.

Comments on Main Development Site

5.50 Our main concerns relate to the methodology and approach to the metric calculations and include:

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483 Draft Development Consent Order (Draft DCO) (AS-143)

484 Draft Development Consent Order (Draft DCO) (AS-143)

485 CIEEM (2018) Guidelines for Ecological Impact Assessment in the UK and Ireland: Terrestrial, Freshwater, Coastal and Marine version 1.1. Chartered Institute of Ecology and Environmental Management, Winchester

486 Draft Development Consent Order (Draft DCO) (AS-143)

- the baseline used is directly at odds with the biodiversity gain system to be introduced by the Environment Bill
- the time for habitats to reach target condition (temporal risk) does not take account of the construction phase
- the % net change in the metric score is calculated versus the on-site baseline but it should be calculated versus the total on-site plus off-site baseline)
- the biodiversity value of existing habitats has not been adequately considered and taken account of
- lack of evidence to support the likelihood of delivering the predicted on-site habitat conditions on post-construction soils
- lack of evidence to support the likelihood of delivering the predicted off-site habitat conditions post development
- the replacement of higher biodiversity[487] habitats with those of lower biodiversity (trading down)

Baseline

5.51 We welcome the inclusion (paragraph 2.3.2 to 2.3.8 of the Biodiversity Net Gain Report[488]) of baseline data dates for Studio Fields complex (on-site and off-site, 2011), Kenton Hills and St James Covert (off-site, 2007), Aldhurst Farm (on-site and off-site, 2015) and the remainder of the red line boundary (on-site, DCO submission). We query the baseline data date for the Marsh Harrier Habitat Improvement Area (on-site and off-site) and as have mentioned above this should be referred to as compensation.

5.52 This baseline policy is directly at odds with the biodiversity gain system set out in the Environment Bill[489], which sets the baseline at the point of the planning application for on-site delivery (Schedule 14, 5(2) as currently drafted) and the point set out in the biodiversity site gain register for off-site delivery (Schedule 10, 1(b) as currently drafted).

> *Schedule 14*
>
> *Pre-development biodiversity value*
>
> *5 (1) In relation to any development for which planning permission is granted, the pre-development biodiversity value of the onsite habitat is the biodiversity value of the onsite habitat on the relevant date.*
> *(2) The relevant date is—*
> *(a) in a case in which planning permission is granted on application, the date of the application, and*
> *(b) in any other case, the date on which the planning permission is granted.*
> *(3) But the person submitting the biodiversity gain plan for approval and the planning authority may agree that the relevant date is to be a date earlier than that specified in sub-paragraph (2)(a) or (b) (but not a date which is before the day on which this Schedule comes into force in relation to the development).*
>
> *Registered offsite biodiversity gains*
>
> *10 (1) "Registered offsite biodiversity gain" means any habitat enhancement, where—*

[487] The metric uses 'distinctiveness' as a collective measure of biodiversity and includes parameters such as species richness, diversity, rarity and the degree to which a habitat supports species rarely found in other habitats.
[488] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)
[489] The Environment Bill, May 2021 - https://bills.parliament.uk/publications/41447/documents/196

(a) the enhancement is required to be carried out under a conservation covenant or planning obligation, and
(b) the enhancement is recorded in the biodiversity gain site register (as to which, see section 93 of the Environment Act 2020).

93 Biodiversity gain site register

(1) The Secretary of State may by regulations make provision for and in relation to a register of biodiversity gain sites ("the biodiversity gain site register").
(2) A biodiversity gain site is land where—
(a) a person is required under a conservation covenant or planning obligation to carry out works for the purpose of habitat enhancement,
(b) that or another person is required to maintain the enhancement for at least 30 years after the completion of those works, and

5.53 The Applicant's policy to apply different on-site baseline data dates to each on-site delivery area (Studio Fields complex, Aldhurt Farm and the marsh harrier compensation area), all of them prior to the point of planning application (paragraph 2.3.2 to 2.3.8 of the Biodiversity Net Gain Report[490]) is at odds with Schedule 5 of the Bill which has no provision for different baseline data dates across the on-site delivery areas. The Applicant has also applied a baseline data date prior to the point of planning application to the off-site delivery areas. As there is currently no biodiversity site gain register, it would not be possible for the Applicant to claim an earlier baseline for off-site delivery under the biodiversity gain system.

The time for habitats to reach target condition (temporal risk)

5.54 Paragraphs 2.5.4 – 2.5.5 of the Biodiversity Net Gain Report[491] consider the temporal risk calculation which is aimed at accounting for the time lag for a newly created or restored habitat parcel to gain target condition, where it is created prior to, or concurrently, with the development. This multiplier is designed to take account of the time between the impact of a development on a habitat and when it is replaced by habitat of equivalent biodiversity value.

5.55 The on-site habitat creation starts post construction, and there will be a period of 8 – 12 years from the point of impact during construction where no habitat will be present. Therefore, the biodiversity unit value of these habitats will be zero, rather than the final figure with a time discount multiplier.

5.56 The Applicant's approach is directly at odds with the biodiversity gain system set out in the Environment Bill[492], which sets the post-development biodiversity value [Schedule 14, 8(1) as currently drafted] as at the time the development is completed.

8 (1) In relation to any development for which planning permission is granted, the post-development biodiversity value of the onsite habitat is the projected value of the onsite habitat as at the time the development is completed.

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490 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

491 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

492 The Environment Bill, May 2021 - https://bills.parliament.uk/publications/41447/documents/196

5.57 Since the on-site habitat creation starts post construction, when the development is completed the value of the on-site habitats will be zero.

5.58 If any habitats are created before the end of development, the temporal risk calculation should consider the time when no habitat is present during construction and should therefore consider the time from the point of impact (when the current habitat is lost) to the estimated time the post construction habitat will reach target quality as noted in the Natural England Biodiversity Metric 2.0 user guide[493] at paragraph 5.19

> *The time period to use in applying the Time to Target Condition multiplier to a metric calculation is the length of time (in years) between the intervention and the point in time the habitat reaches the pre-agreed target quality (i.e. distinctiveness, condition, area). This time will vary between habitat types, between change scenarios (e.g. creation typically takes longer than enhancement) and due to way the habitat is managed. Time to target condition values – based on based on good practice and typical conditions are provided for all habitats used in biodiversity metric 2.0. These values are set out in detail in the Technical Supplement.*

5.59 This suggests that more offsite habitat creation should be completed to compensate for temporal losses of biodiversity in line with the Natural England Biodiversity Metric 2.0 user guide[494] which notes at paragraph 5.10

> *A typical consequence of applying risk multipliers is to increase the size (e.g. area or length for linear features) of habitat required as compensation such that it exceeds the size of habitat lost or damaged. This is necessary:*
>
> *to compensate for temporal losses of biodiversity (e.g. where there is a period of diminished biodiversity between the point in time when a habitat is impacted and it is replaced by habitat of equivalent biodiversity value);*

5.60 In addition, we recommend all habitat creation commences as soon as possible and we query whether there may be opportunities to transplant more mature trees to reduce time to reach target condition.

5.61 When considering the off-site habitat creation, the Defra Biodiversity Offsetting Pilots guidance[495] notes at paragraph 69

> *The number of years that time discounting should take into consideration is from the point of impact to the estimated time that it will take for the habitat to reach the pre-agreed target quality (i.e. the point at which the agreed number of units is delivered).*

5.62 Where off-site habitat creation starts after the impact occurs the temporal risk calculation should also consider the time from the point of impact to the estimated time to reach target quality.

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493 Crosher, I. A, Gold, S. B., Heaver M.D., Heydon M.A., Moore L.D., Panks, S.A., Scott S.C., Stone D.A. and White N.A. (2019) The Biodiversity Metric 2.0: auditing and accounting for biodiversity value. User guide (Beta Version, July 2019). Natural England

494 Crosher, I. A, Gold, S. B., Heaver M.D., Heydon M.A., Moore L.D., Panks, S.A., Scott S.C., Stone D.A. and White N.A. (2019) The Biodiversity Metric 2.0: auditing and accounting for biodiversity value. User guide (Beta Version, July 2019). Natural England

495 Defra (March 2012) Biodiversity Offsetting Pilots Technical Paper: the metric for the biodiversity offsetting pilot in England

Net % change calculation

5.63 The Biodiversity Net Gain Report[496] at section 8.1.1 image 4 summarises the results of the assessment and reports 18.03% net change in the metric score. Using the figures provided there, the reported 18.03 % net change is the change in the metric score versus the on-site baseline as demonstrated below.

> Total baseline biodiversity units = on site 1244.45 + off-site 429.99 = 1,674.44
>
> Total post intervention biodiversity units = on site 918.51 + off-site 980.27 = 1,898.78
>
> Total net unit change = 1,898.78-1,674.44= 224.34
>
> Total net % change vs. on site baseline = 224.34/1244.45*100 = 18.03%
>
> Total net % change vs. total baseline = 224.34/ 1,674.44*100 = 13.40%

5.64 We note it is necessary for the net % change to be calculated versus the total baseline (on-site plus off-site baseline). In addition, any off-site habitat creation that requires planning permission would need to deliver an additional 10% net gain units in order to meet its own requirements under the biodiversity gain system.

Difficulty of creating or restoring a habitat

5.65 Paragraph 2.5.1 of the Biodiversity Net Gain Report[497] considers the difficulty of creating or restoring a habitat. We request confirmation that the recommended standard difficulty scores are used for each habitat type.

Evidence base for on-site habitat creation

5.66 We have concerns over the evidence base to support the likelihood of achieving the target on-site habitat conditions post construction, in particular the target 'lowland dry acid grassland' of 'fairly good' condition in most areas of the site.

5.67 Paragraphs 4.1.26 – 4.1.32 of the Biodiversity Net Gain Report[498] discuss creation of dry Sandlings grassland (DSG) on the construction site post construction.

5.68 Paragraph 4.1.27 notes uncertainty over the future nature of post-construction soils

> *Some flexibility is required for the creation and management of this area, since there is some uncertainty over the future nature of the post-construction soils*

and

> *The ultimate vision for the DSG (summer parched grassland) is that these develop over time into an acid grassland / heathland mosaics depending on the realities of soil pH and structure and water availability.*

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[496] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[497] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[498] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

5.69 Paragraph 4.1.29 notes reversion of arable to acidic grassland at Aldhurst Farm can indicate future habitat creation outcomes specifically creation of 'fairly good' condition lowland dry acid grassland

> *It is considered to be achievable to target a 'Lowland dry acid grassland' sward in most areas of the site. Actions to revert arable habitat to an acidic grassland sward is already underway within Aldhurst Farm and the Studio Fields complex. These areas are off-site, but provide an indication of what can be achieved and can be learned from for future habitat creation interventions. A species-rich sward will be targeted, with wildflower cover around 30%. Positive management will target achievement of the condition assessment criteria. As such, a condition of 'fairly good' is predicted. This reflects the challenges of creating areas of such habitat and remains realistic, while targeting a positive outcome.*

5.70 We have concerns around using habitat creation from arable land at Aldhurst Farm and Studio Fields to predict creation of dry Sandlings grassland of fairly good condition on land impacted by construction.

5.71 Paragraph 5.3.28 of the Biodiversity Net Gain Report[499] discusses habitat creation in the marsh harrier compensation area and predicts creation of fairly good condition lowland dry acid grassland and fairly good condition lowland heathland from former arable land. It seems unlikely that a similar outcome will be achieved on post construction soils.

5.72 Sizewell B was constructed from 1988 to 1995[500]. Paragraphs 3.1.46 - 3.4.47 of the Biodiversity Net Gain Report[501] note the impact of Sizewell B construction

> *3.1.46 A large area of semi-improved neutral grassland is present to the east of the main platform. This area of land was impacted by construction processes associated with the Sizewell B construction works. As a result, the habitat does not have a typical dune grassland character, due to changes to the hydrology and pedology[502] caused by the construction works.*
>
> *3.1.47 The habitat to the east of the main platform is of semi-improved character, dominated by Cock's-foot and common herbs. Undesirable species and scrub are present across the area. As a result, this habitat is considered to be of moderate condition.*

5.73 This assessment concludes that land impacted by the construction of Sizewell B suffered changes in hydrology and pedology[503] and some 25 years later is semi-improved neutral grassland of moderate condition. It seems reasonable to conclude that land impacted by the 8-12 year Sizewell C construction period is likely to suffer changes that will affect the proposed habitat creation. It would be more precautionary to use habitat creation outcomes on the Sizewell B construction site as a predictor of future habitat condition on the Sizewell C construction site instead of using outcomes from habitat creation at Aldhurst Farm.

5.74 The predicted metric score is dependent on successfully creating target habitat conditions. We request evidence is provided to support the likelihood of delivering the predicted habitat conditions on land impacted by Sizewell C construction post development. We consider the target condition should take account of the difficulty of habitat creation on post construction soils.

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499 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

500 Sizewell B power station

501 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

502 Physical and chemical properties of soil

503 Physical and chemical properties of soil

Evidence base for off-site habitat creation

5.75 We also have concerns over the evidence base to support the likelihood of achieving the target off-site habitat conditions.

5.76 Paragraphs 5.3.9 - 5.3.10 of the Biodiversity Net Gain Report[504] discuss creation of acid grassland at Aldhurst Farm and notes challenges achieving the targeted level of diversity

> *it has been challenging to achieving the targeted level of diversity. As such, a precautionary approach is taken and it is assumed that the highest distinctiveness typology will not be achieved. The 'other lowland acid grassland' typology is therefore predicted. It is predicted that good condition can be achieved, through the positive management outlined in the Aldhurst Farm ELMP and due to a lower distinctiveness typology predicted.*

5.77 Paragraph 5.3.28 discusses habitat creation in the Marsh Harrier Habitat Mitigation Area and predicts creation of fairly good condition lowland dry acid grassland and fairly good condition lowland heathland from former arable land.

5.78 The predicted BNG is dependent on successfully creating the target habitat conditions. We request evidence is provided to support the likelihood of delivering the predicted off-site habitat conditions post development.

Use of 'fairly good' category

5.79 Table 13 of the Biodiversity Net Gain Report[505] notes dry Sandlings grassland of 'fairly good' condition provides 278.66 biodiversity units out of a total of 758.76 units for the on-site areas. The new metric due to be released in Summer 2021 will not include the 'fairly good' category and the condition will consequently need to be changed to moderate which will reduce the assessment for dry Sandlings grassland to 222.93 units.

5.80 Table 22 of the Biodiversity Net Gain Report[506] notes heathland and scrub, woodland forest and grassland of 'fairly good' condition provide 123.33 biodiversity units out of a total of 980.72 units for the off-site areas. The new metric due to be released in Summer 2021 will not include the 'fairly good' category and the condition will consequently need to be changed to moderate which will reduce the assessments for those habitats to 98.66 units.

Replacement of higher distinctiveness habitats with those of lower distinctiveness

5.81 The metric uses distinctiveness as a collective measure of biodiversity and includes parameters such as species richness, diversity, rarity and the degree to which a habitat supports species rarely found in other habitats.

5.82 Table 19 of the Biodiversity Net Gain Report[507] shows huge reductions in total biodiversity unit values of high distinctiveness habitat: sparsely vegetated coastal habitats (-94%) and woodland (-

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[504] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[505] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[506] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

[507] ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

40%). The Defra Biodiversity Offsetting Pilots guidance [508] states at paragraph 22 that habitats of high distinctiveness would be expected to be offset with 'like for like'

> *22. One of the guiding principles for developing our approach to offsetting is that it should result in an improvement in the extent or condition of the ecological network. To do this the focus of habitat restoration or creation through offsetting should be on priority habitats. Where development is taking place on habitats in the low distinctiveness band, the offset actions should result in expansion or restoration of habitats in the medium or, preferably, high distinctiveness band. At no time should an offset result in "trading down", for instance in the replacement of habitat of high distinctiveness with creation or restoration of a habitat of medium distinctiveness. Habitats that are of high distinctiveness would generally be expected to be offset with "like for like" i.e. the compensation should involve the same habitat as was lost.*

5.83 It would be expected that habitats of high distinctiveness (such as sparsely vegetated coastal habitats and some woodland types) would not see an overall reduction in habitat unit value. In the calculations, the significant increase in grassland habitats is compensating for a loss in these habitats.

5.84 The Natural England Biodiversity Metric 2.0 user guide[509] at rule 3 (section 2.23) notes trading down must be avoided

> *'Trading down' must be avoided. Losses of habitat are to be compensated for on a 'like for like' or 'like for better' basis. Ideally, new or restored habitats should aim to achieve a higher distinctiveness or condition than habitats lost.*

5.85 And at rule 5 (section 2.23) notes it may be necessary to enhance or create a larger area of habitat than lost to fully compensate for impacts on biodiversity

> *It is not the area of habitat created that determines whether ecological equivalence or better has been achieved but the net change in biodiversity units. Risks associated with enhancing or creating habitats mean that it may be necessary to enhance or create a larger area of habitat than lost to fully compensate for impacts on biodiversity.*

5.86 BNG Good Practice Principles for Development[510] notes at section 11.4.1

> *Good practice is achieving net gains of the same or higher biodiversity value as those affected and equivalent or better levels of ecological functionality.*

5.87 More habitat creation is therefore required to fully compensate for impacts on biodiversity. If sparsely vegetated coastal habitats cannot be created on-site, then off-site enhancements would be expected. In addition to the reduction in overall biodiversity unit value of woodland, semi-natural broadleaved woodland is largely being offset by mixed woodland. It would be expected that habitats of high distinctiveness (such as semi-natural broadleaved woodland) are replaced like for like as expected in paragraph 22 of the Defra Biodiversity Offsetting Pilots guidance[511]

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508 Defra (March 2012) Biodiversity Offsetting Pilots Technical Paper: the metric for the biodiversity offsetting pilot in England

509 Crosher, I. A, Gold, S. B., Heaver M.D., Heydon M.A., Moore L.D., Panks, S.A., Scott S.C., Stone D.A. and White N.A. (2019) The Biodiversity Metric 2.0: auditing and accounting for biodiversity value. User guide (Beta Version, July 2019). Natural England

510 Baker, J., Hoskin, R. and Butterworth, T. (2019) Biodiversity net gain. Good Practice Principles for Development. A practical guide.

511 Defra (March 2012) Biodiversity Offsetting Pilots Technical Paper: the metric for the biodiversity offsetting pilot in England

Habitats that are of high distinctiveness would generally be expected to be offset with "like for like" i.e. the compensation should involve the same habitat as was lost.

Hedgerows

5.88 Paragraph 2.2.7 of the Biodiversity Net Gain Report[512] notes hedgerows were grouped together for assessment and it is not clear if each hedgerow was surveyed for the condition assessment. We are therefore concerned hedgerows may have been undervalued and we request clarification of the evidence base for hedgerow condition assessment.

5.89 Although a hedgerow is a linear feature, it does occupy an area and we query whether there may be double counting of hedgerow and habitat areas. We request confirmation that the area of hedgerows has been removed from the area of adjacent habitat parcels.

5.90 The on-site hedgerow baseline biodiversity units reported in section 4.1.5 of the Biodiversity Net Gain Report[513] is different to the value in Table 14.

Areas excluded from the assessment

5.91 All habitat improvements funded through Sizewell B commitments should be excluded from the Sizewell C metric score calculation and we welcome the exclusion of the reversion to grassland at Retsom's Field (Sizewell B commitment) from the updated assessment (paragraph 2.10.6 of the Biodiversity Net Gain Report[514]).

5.92 We also welcome the exclusion of the Sizewell Estate Woodland Management Plan that defines the management of the existing woodlands within the site and the wider EDF Energy estate from the assessment. But Studio Fields Complex, included within the EDF Energy Estate Pre-Construction Integrated Land Management Plan, remains (Table 3.1 of the Biodiversity Net Gain Report[515]). Paragraph 3.4.1 of the oLEMP[516] helpfully explains

Management of these areas would be continued as defined by those existing plans and therefore management proposals for these areas are not considered further within this oLEMP.

5.93 As mentioned above the oLEMP[517] paragraph 4.1.7 notes

The Biodiversity Net Gain report …. demonstrates that net gain would be achieved if the habitats described within this oLEMP are delivered as proposed.

5.94 We therefore request clarification that the measures at Studio Fields Complex included in the metric calculation are additional to those in the existing management plan.

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512 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

513 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

514 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

515 ES Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14E Biodiversity Net Gain Report (REP1-004)

516 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)

517 8.2 Outline Landscape and Ecology Management Plan (oLEMP) (REP1-010)

Comments on two village bypass

Baseline conditions and valuation

5.95 The baseline data date is not provided, and we assume it to be at the time of DCO submission i.e. 2020. We request confirmation of the baseline data date for the two village bypass.

5.96 Paragraph 2.11.1 of Biodiversity Net Gain Report Two Village Bypass[518] notes

> *Arcadis have used third party data as part of the assessments of the post development and off-site habitats and assumptions on the condition of the baseline habitats are inferred from existing data.*
>
> *No specific surveys or assessments were undertaken.*

5.97 We request evidence to support the baseline condition assessments. There should be sufficient surveys to ensure a demonstrable 10% increase in the metric score.

5.98 Paragraph 3.1.7 of Biodiversity Net Gain Report Two Village Bypass[519] considers the condition of semi-natural broad-leaved woodland to be lost and notes

> *The woodlands are all relatively young and comprised mainly native species such as Beech (Fagus sylvatica), Hazel (Corylus avellana) and Field Maple (Acer campestre). Locally a well-developed ground-layer with species such as Dog's Mercury (Mercurialis perennis) and Bluebell (Hyacinthoides non-scripta). The age structure is not diverse enough to classify the woods as good condition, but the precautionary method was used and fairly good condition was selected.*

5.99 Field maple, dog's mercury and bluebell are recognised as ancient woodland indicator plants in the east of England[520]. Their presence on site does suggest the semi-natural broadleaved woodland is of value and a precautionary approach would be to select good condition for habitat to be lost.

Habitat creation

5.100 The predicted metric score is dependent on achieving the target habitat conditions. Paragraph 3.2.23 onwards of Biodiversity Net Gain Report Two Village Bypass[521] note proposed broadleaved woodland, neutral grassland, lowland acid grassland, infiltration basins and ponds will achieve good condition. We request evidence is provided to support the likelihood of delivering the predicted habitat conditions.

5.101 If is important to note that the time to target condition is 15 years for the grassland habitats and 32+ years for woodland. In addition we have concerns over long-term management to achieve these target conditions. The Two Village Bypass Outline Landscape and Ecology Management Plan[522] notes at paragraph 1.1.6

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518 9.6 Biodiversity Net Gain Report - Two Village Bypass (REP1-018)
519 9.6 Biodiversity Net Gain Report - Two Village Bypass (REP1-018)
520 Table of Ancient Woodland Indicator Plants in Rose, F. (2006) The Wild Flower Key. Warne
521 9.6 Biodiversity Net Gain Report - Two Village Bypass (REP1-018)
522 Associated Development Design Principles: Appendix A: Two Village Bypass Outline Landscape and Ecology Management Plan (AS-263)

The oLEMP and subsequent LEMP would be managed by SZC Co. for a total of five years, or until adoption by the Highways Authority. It is expected that the LEMP would contain long term management objectives for the proposed development.

5.102 We are concerned that Schedule 2 of the Draft DCO[523] does not include a specific requirement for long-term management of the habitats required for securing a 10% metric score in accordance with planning policy.

5.103 We also query whether there may be opportunities for net gain habitat enhancements to provide habitat for protected species of bats and birds. The appendices are missing from the report. We will review and comment on the baseline Phase 1 map and operational masterplans when they are submitted to the Examination. We hope the Application will include an innovatively designed green bridge linking Foxburrow Wood County Wildlife Site with the woodland to the west.

Comments on Sizewell Link Road

Baseline conditions and valuation

5.104 The baseline data date is not provided, and we assume it to be at the time of DCO submission i.e. 2020. We request confirmation of the baseline date for the Sizewell Link Road.

5.105 Paragraph 2.10.1 of Biodiversity Net Gain Report Sizewell Link Road[524] notes

Arcadis have used third party data as part of the assessments of the post development and off-site habitats and assumptions on the condition of the baseline habitats are inferred from existing data.

No specific surveys or assessments were undertaken.

5.106 We request evidence to support the baseline condition assessments. There should be sufficient surveys to ensure a demonstrable 10% increase in the metric score.

5.107 Table 8 of Biodiversity Net Gain Report Sizewell Link Road[525] describes minor changes were made to the site boundary since the previous Biodiversity Net Gain report. Many of the post development habitat descriptions use the phrase 'appear to be'. We request clarification of the post development habitats.

Habitat creation

5.108 The predicted metric score is dependent on achieving the target habitat conditions. Paragraph 3.2.22 onwards of Biodiversity Net Gain Report Sizewell Link Road[526] note proposed broadleaved woodland, neutral grassland, infiltration basins and ponds will achieve good condition. We request evidence is provided to support the likelihood of delivering the predicted habitat conditions. The time to target condition is 15 years for the grassland habitats and 32+ years for woodland and we also have concerns over long-term management to achieve the target conditions.

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523 Draft Development Consent Order (Draft DCO) (AS-143)
524 9.5 Biodiversity Net Gain Report - Sizewell Link Road (REP-017)
525 9.5 Biodiversity Net Gain Report - Sizewell Link Road (REP-017)
526 9.5 Biodiversity Net Gain Report - Sizewell Link Road (REP-017)

5.109 We are concerned that Schedule 2 of the Draft DCO[527] does not include a specific requirement for long-term management of the habitats and securing a 10% net gain score in accordance with planning policy.

5.110 The appendices are missing from the report. We will review and comment on the baseline Phase 1 map and operational masterplans when they are submitted to the Examination.

Conclusions

5.111 In conclusion, based on our above concerns and the information available at present, we do not agree the Application can achieve net gain due to direct adverse impact on Sizewell Marshes SSSI from loss of a significant proportion of the SSSI, nor the amount claimed by the Applicant.

5.112 Despite this, we agree the Application should be meeting the requirements of the net gain system and should be providing landscape scale net gain proportionate to the scale of the project in line with planning policy. As such, the Application should be demonstrating a high metric score that does not rely on protected species mitigation and/or compensation habitat to get over the 10% uplift requirement.

5.113 Our key concerns relating to methodology and approach to the metric calculations include:

- use of measures that are for protected sites and species mitigation and compensation
- the baseline policy is directly at odds with the biodiversity gain system to be introduced by the Environment Bill
- the time for habitats to reach target condition does not take account of the construction phase
- the replacement of higher biodiversity habitats with those of lower biodiversity
- lack of evidence to support the likelihood of delivering the predicted habitat conditions
- the biodiversity value of existing habitats has not been adequately considered and taken account of
- heavy use of 'fairly good' category
- the % net change in the metric score is calculated versus the on-site baseline but it should be calculated versus the total on-site plus off-site baseline

5.114 We consider the key finding from the Applicant's BNG proposals to be at best only just over the 10% boundary by a narrow margin, and that if any one of several marginal decisions were reversed it would fall below it. We therefore conclude the Applicant will fail to meet the claimed BNG.

6. LAND AND SOCIO-ECONOMIC CONCERNS

Land

6.1 The RSPB owns land immediately adjoining the development site. The RSPB has been in discussions on the precise alignment of this boundary which is physically different to the existing fence lines on the ground. Whilst there is no specific dispute on the legal boundary, we are unclear how any new boundary will appear physically and as such how that will impact on our boundary. We are not clear on how for example the Sandlings Footpath (permissive) will be re-routed. The RSPB reserves its position on the precise boundary until these plans are made clear.

6.2 The RSPB manages RSPB Minsmere both as a nature reserve and as agricultural holding with livestock. With a development of this scale, we are concerned about the impact on the

[527] Draft Development Consent Order (Draft DCO) (AS-143)

management of our holding, whether that be through disturbance from noise, light, dust or air pollution.

6.3 We have previously worked with the Applicant to allow access to the RSPB Minsmere to carry out monitoring for the purpose of informing the Application, but we are concerned that there is no provision for monitoring for matters such as recreational displacement, coastal processes, hydrological monitoring or other ecological data currently to enable the Applicant to review assessments done and to check whether the proposed development has further, unpredicted, impacts.

Resilience Fund and Other Proposed Section 106 Funds

6.4 The RSPB is grateful to the Applicant for its proposals for a Resilience Fund for the RSPB (and others). As we understand it, this fund will be to off-set perceived impacts and risks caused by the proposed development. We have to date had outline discussions with the Applicant but the further detailed required are yet to be worked up.

6.5 We would seek to ensure that the Resilience Fund is available for the full period of the construction programme. But also are keen understand more about how precisely the Resilience Fund would practically operate.

6.6 As mentioned above the RSPB does considers that the proposed development will have impacts on visitors to RSPB Minsmere and indeed the wider area and whilst a Resilience Fund is welcome, we believe there is little evidence in the Application documents as to any acknowledgement as to how any consequential loss will be addressed.

6.7 Again as mentioned above we would also like to see additional survey work undertaken throughout the construction period on visitors both directly at RSPB Minsmere and across the wider area and to ensure the impacts of the development can monitored.

6.8 Due to its location, the proposed development will have a significant visual impact on visitor experience and the views available, with increases in noise and light pollution havin an effect on the tranquillity and character of the landscape which makes RSPB Minsmere such a special place for our visitors. We do not agree with the Applicant's position that, impacts on the landscape and visual effects are only localised, across the AONB.

6.9 For example in the Landscape and Visual Impact Assessment, the Applicant does recognises this impact but state it would localised across the AONB and the effects are not significant. The RSPB disagrees with this conclusion. The RSPB has invested heavily in its business and the visual and other impacts on visitor experience are likely to have an impact on its business. Currently we do not believe the Resilence Fund is designed to address these impacts.

6.10 Additionally, there is a concern that by agreeing to accept funding as part of a Resilience Fund that this would prevent us from being about to access other appropriate funds for mitigation for other issues measures that may be required for ongoing monitoring. It may also preclude us from being able to work in partnership with others to deliver mitigation beyond our site boundary.

6.11 We are continuing our discussions with the Applicant on these issues and concerns and will update the Examining Authority during the course of the Examination.

Second Leiston Abbey Farm Site

6.12 The RSPB is also grateful for provisions being made towards the Second Leiston Abbey Farm Site Enhancement Scheme (as referred in the draft Section 106, Schedule 8). The RSPB has not been

part of any detailed discussions surrounding this enhancement Scheme and as landowner and due to its statutory duty to maintain the Scheduled Ancient Monument we would welcome discussions on how the Scheme will operate and what enhancement measures may be possible within it. as far as we are aware these details and proposals are not currently before the Examination.

7. CONCLUSIONS

7.1 In conclusion, based on the above concerns, we do not agree that *adverse effects on integrity* can be excluded for the following sites and species for impacts from the Application alone:

- Breeding gadwall, shoveler, teal, marsh harrier, nightjar, avocet, bittern and little tern and non-breeding white-fronted goose, gadwall, shoveler and hen harrier of the Minsmere-Walberswick SPA
- Non-breeding red-throated divers of the Outer Thames Estuary SPA
- Breeding Sandwich terns of the Alde-Ore Estuary SPA
- Perennial vegetation of stony banks and European dry heaths of the Minsmere-Walberswick Heaths & Marshes SAC
- The Ramsar site - specifically criterion 1 - mosaic of marine, freshwater, marshland and associated habitats, complete with transition areas in between, criterion 2 nine nationally scarce plants and at least 26 red data book invertebrates, and the important assemblage of rare breeding birds associated with marshland and reedbeds

7.2 In relation to the Application in combination with other plans and projects, we do not agree that *adverse effects on integrity* can be excluded for the following sites and species:

- Breeding gadwall and shoveler, nightjar and little tern and non-breeding white-fronted goose, gadwall, shoveler, hen harrier of the Minsmere-Walberswick SPA
- Non-breeding red-throated divers of the Outer Thames Estuary SPA
- Breeding woodlark and nightjar of the Sandlings SPA
- Perennial vegetation of stony banks and European dry heaths of the Minsmere-Walberswick Heaths & Marshes SAC
- The Ramsar site - specifically criterion 1 - mosaic of marine, freshwater, marshland and associated habitats, complete with transition areas in between, criterion 2 nine nationally scarce plants and at least 26 red data book invertebrates, and the important assemblage of rare breeding birds associated with marshland and reedbeds

7.3 We also consider that significant impacts from the Application alone are likely on the following sites:

- Minsmere-Walberswick Heaths and Marshes SSSI
- Sizewell Marshes SSSI

7.4 We also consider that significant impacts from the Application alone are likely on the following species:

- Bats, in particular barbastelle, Natterer's
- Natterjack toad
- stone curlew
- Dartford warbler

7.5 We will continue to work with the Applicant on these concerns and provide comments on further information submitted to the Examination for example further details on the required management mitigation, compensation and monitoring plans and schemes as well as the details of the various funds mentioned above. We will update the Examining Authority of our position in light of any new information presented

7.6 However for now (as also covered in other section of these written representations) we have serious concerns about the adequately of information currently available and strongly recommend that without those details including the lack of clarity on the required legal, financial and ecological securing of requirements, measures, plans and schemes – confidence in them is premature.

8. THE LOVE MINSMERE CAMPAIGN

What is the Love Minsmere campaign?

8.1 RSPB Minsmere nature reserve is not only incredibly important to the 6000 types of birds, animals, plants, and fungi that call it home, but also the people that visit in huge numbers[528] every year to enjoy this special place.

8.2 In 2019, the RSPB was increasingly concerned about the potentially damaging impacts of the Application on RSPB Minsmere and the wildlife that lives there. The RSPB decided to create a public campaign that would allow its supporters to show that they Love Minsmere and that they want to see this incredible home for wildlife protected.

8.3 In 2020 Suffolk Wildlife Trust joined forces with the RSPB in the Love Minsmere campaign, which now campaigns to see both RSPB Minsmere nature reserve, the protected and designated sites including Sizewell Marshes SSSI (which Suffolk Wildlife Trust manage), and the surrounding wildlife that calls the Suffolk coast home, protected against potentially harmful impacts from the Application.

8.4 To date the Love Minsmere campaign has comprised of three different actions as explained in the following sections.

Stage 3 Public Consultation

8.5 During the Stage 3 public consultation in 2019, the RSPB asked supporters to submit consultation responses to the Applicant telling them why they Love Minsmere and why it must be protected. In these responses they also called on the Applicant to publicly pledge to protect RSPB Minsmere and the wildlife that calls it home.

8.6 Over the course of the consultation, 20,419 supporters submitted consultation responses to the Applicant on the importance of RSPB Minsmere and why it must be protected.

The Love Minsmere Festival

8.7 Later in 2019, on the 15 September, during the Application's fourth and final consultation, the RSPB held their first ever Love Minsmere Festival. The Love Minsmere Festival was a day to celebrate how special RSPB Minsmere is for people and wildlife. It was also an opportunity to show the Applicant once again how many people Love Minsmere and wish to see it protected.

[528] From our regular monitoring of visitor numbers we know that we have had up to 123,966 visitors per year, with an average number of visitors in the past 5 non-covid years of 110,649 visitors per year.

8.8 Over 1000 people gathered on Whin Hill, the site of the BBC Springwatch studio[529]2, and one of the reserve's most amazing viewpoints, and together they collectively declared their love for Minsmere. Supporters formed the outline of the Love Minsmere love heart logo, with Sizewell in the background, sending a visual message to the Applicant that the reserve must be protected.

Figure 2. An image of over 1000 people forming a human Love Minsmere love heart at RSPB Minsmere nature reserve as part of the Love Minsmere Festival, 15 September 2019.



Pre-examination e-action

8.9 Ahead of the formal start of the Examination, the RSPB and Suffolk Wildlife Trust came together to launch the next leg of the ongoing Love Minsmere campaign[530].

8.10 Between the 27 November 2020 and the 15 February 2021, both organisations ran a Love Minsmere e-action[531]. We called on supporters who shared our summary of concerns[532] (based on our Relevant Representations[533]) regarding the Application's potential negative impacts on

[529] BBC Springwatch was based at RSPB Minsmere nature reserve in 2014, 2015 and 2016
[530] For our latest campaign news see

[531] The Love Minsmere e-action was hosted online a website. It enabled people to add their name and address if they shared our summary of concerns (based on our Relevant Representations) regarding the Application's potential negative impacts on nature, thereby signing our e-action

[532] The RSPB and Suffolk Wildlife Trust's shared summary of concerns can be found here: https://www.rspb.org.uk/globalassets/downloads/sizewell/swt-and-rspb-shared-concerns-for-sizewell-c.pdf
[533] https://www.rspb.org.uk/globalassets/images/campaigning-and-positions/love-minsmere/rspb-relevant-representations-sizewell-c.pdf

nature, to sign our e-action and demonstrate how many people want to see RSPB Minsmere nature reserve, the protected and designated sites including Sizewell Marshes SSSI as well as the surrounding wildlife of the Suffolk coast protected.

8.11 In just 12 weeks, 104,836 people took our e-action and said they share our concerns over the Application's potential harmful impacts on nature. We told these 104,836 people that we would represent their concerns in the examination by including reference to the e-action and the number of people that shared our concerns in our Written Representations.

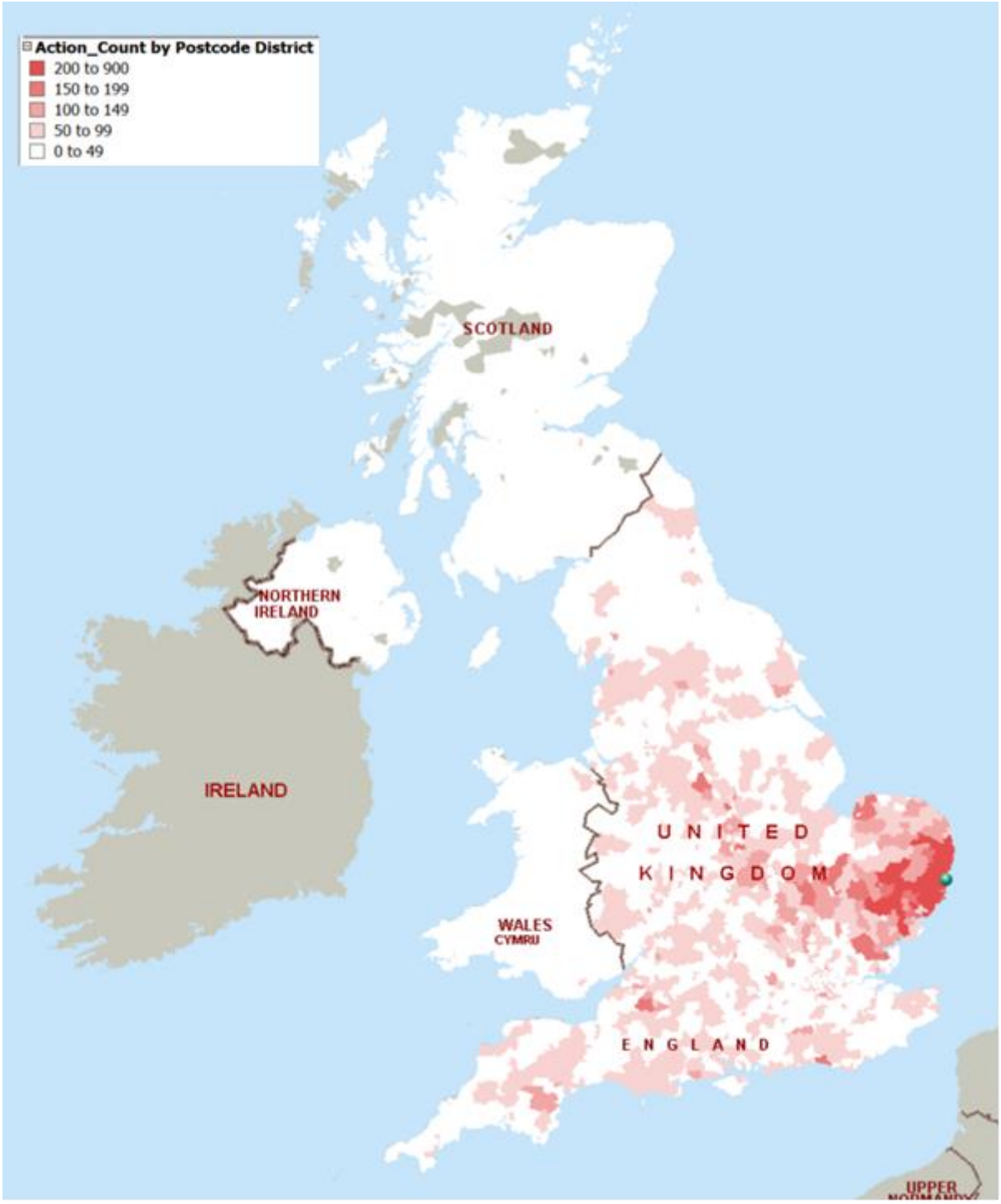
8.12 The number of people signing the Love Minsmere e-action has eclipsed the number of signatures and responses to all previous Sizewell C consultations and campaigns. It may also cast doubt over claimed support for Sizewell C.

8.13 The majority of our supporters who signed the Love Minsmere e-action were located on the Suffolk coast demonstrating the concern amongst local people for their local wildlife, but we saw supporters sign the e-action across the UK showing how many people Love Minsmere across the country and are concerned about the harmful impacts the Application could have on the birds, animals, plants and the ecosystems they call home.

8.14 Locally, in the Suffolk Coastal constituency[534], home to both RSPB Minsmere nature reserve, Sizewell Marshes SSSI and the Application site, 3,436 people signed the e-action – the highest number across all of the constituencies. Across the county of Suffolk this number grew with over 10,000 people sharing our concerns. However, the location of some of the people that signed the e-action is unknown so these figures could be higher.

[534] We were able to break down the location of our signatories by both parliamentary constituency and postcode district

Figure 2. A heat map showing by volume the distribution of the Love Minsmere actions (between 27 November 2020 – 15 February 2021) by Postcode District:



8.15 Support for the Love Minsmere campaign is national, not just local, and we have also seen public support from a whole host of celebrities including Chris Packham, Anthony Horowitz, Julia Bradbury, Alison Steadman, Samuel West, Deborah Meaden, Bill Turnbull, Michaela Strachan, Mike Dilger, Megan McCubbin Iolo Williams, Emma Kennedy, Miranda Krestovinkoff, Dr Amir Khan and more.

8.16 We believe that the 104,836 people signing our most recent Love Minsmere e-action, and the support for our previous campaign actions, shows just how special RSPB Minsmere, Sizewell

Marshes SSSI and the Suffolk coast is for both people, and wildlife. We also believe it shows the breadth and strength of concern regarding the Application and potential impacts on nature.

Appendices
to
Written Submission

for the
Royal Society for the Protection of Birds
and Suffolk Wildlife Trust

Submitted for Deadline 2
2 June 2021

Planning Act 2008 (as amended)

In the matter of:

Application by NNB Generation Company (SZC) Limited for an Order Granting Development Consent for

The Sizewell C Project

Planning Inspectorate Ref: EN010012
RSPB Registration Identification Ref: 20026628
Suffolk Wildlife Trust Registration Identification Ref: 20026359

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Appendix 1: Species Accounts

Appendix 2: Footprint Ecology reports regarding recreational impacts:

- Liley, D. & Saunders, P. (2020a) Impacts of recreation related to Sizewell C and implications for European sites. Unpublished report by Footprint Ecology
- Liley, D. & Saunders, P. (2020b) Review of Sizewell C application documents and evidence in relation to recreation impacts. Unpublished report by Footprint Ecology.

Appendix 3: Natterjack toad report:

- Amphibian and Reptile Conservation (2020). Sizewell C Project: comments on natterjack toad mitigation proposals. Report to Suffolk Wildlife Trust. ARC report. ARC, Bournemouth.

Appendix 4: List of References

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1.1 Greater white-fronted goose (*Anser albifrons albifrons*)





Key Information: The white-fronted goose is a grey goose, bigger than a mallard and smaller than a mute swan. Adults have a large white patch at the front of the head around the beak and bold black bars on the belly. The legs are orange and Siberian birds have pink bills, while Greenland birds have orange bills. This species does not breed in the UK. Two subspecies visit the UK in winter - birds which breed in Greenland (*Anser albifrons flavirostris*) and birds which breed in Siberia (*Anser albifrons albifrons*); in the East of England the Siberian subspecies occurs. The current wintering areas need protection, including avoiding drainage of traditional wintering areas in southern England.

What they eat: Grass, clover, grain, winter wheat and potatoes.

UK conservation status[1]: Red

UK Wintering (Individual): 2,100[2]

Importance level: 0.18-0.21% of the biogeographic population (NW Siberia & NE/NE Europe) winters in UK

1.2 Shoveler (*Spatula clypeata*)





Key Information: Shovelers are surface feeing ducks with huge spatulate bills. Males have dark green heads, with white breasts and chestnut flanks. Females are mottled brown. In flight birds show patches of light blue and green on their wings. In the UK they breed in southern and eastern England, especially around the Ouse Washes, the Humber and the North Kent Marshes and in much smaller numbers in Scotland and western parts of England. In winter, breeding birds move south, and are replaced by an influx of continental birds from further north. The UK is home to more than 20 per cent of the NW European population.

What they eat: Small insects and plant matter sifted from the water.

UK conservation status[3]: Amber

UK Wintering (Individual): 19,500[4] **UK Breeding (Pairs):** 1,100[5]

Importance level: 30% of the biogeographic population (NW and Central Europe) winters in UK
3.38% of the biogeographic population breeds in UK

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1 Eaton, M., Aebischer, N., Brown, A., Hearn, R., Lock, L., Musgrove, A., Noble, D., Stroud, D. & Gregory, R. (2015) Birds of Conservation Concern 4: the population status of birds in the UK, Channel Islands and Isle of Man (App XX, Tab XX)

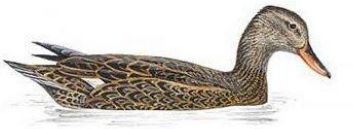
2 Woodward, I., Aebischer, N., Burnell, D., Eaton, M., Frost, T., Hall, C., Stroud, D.A. & Noble, D. (2020). Population estimates of birds in Great Britain and the United Kingdom. Summary. Full Report: British Birds 113: 69–104 (App XX, Tab XX)

3 Eaton et al (2015). See footnote 1

4 Woodward et al (2020). See footnote 2

5 Woodward et al (2020). See footnote 2

1.3 Gadwall (*Mareca strepera*)





Key Information: The gadwall is a very grey-coloured dabbling duck, a little smaller than the mallard, and with an obvious black rear end. It shows a white wing patch in flight. When seen close up the grey colour is made up of exquisitely fine barring and speckling. It nests in low numbers in the UK.

What they eat: Stems, leaves and seeds.

UK conservation status[6]: Amber

UK Wintering (Individual): 31,000[7] **UK Breeding (Pairs):** (1,250-3,200)[+8]

Importance level: 26% of biogeographic population (NW Europe) winters in UK
1.8 – 5.7% of the biogeographic population breeds in UK

1.4 Teal (*Anas crecca*)





Key Information: Teals are small dabbling ducks. Males have chestnut coloured heads with broad green eye-patches, a spotted chest, grey flanks and a black edged yellow tail. Females are mottled brown. Both show bright green wing patches (speculum) in flight. They are thinly distributed as a breeding species with a preference for northern moors and mires.

In winter, birds congregate in low-lying wetlands in the south and west of the UK. Of these, many are continental birds from around the Baltic and Siberia. At this time, the UK is home to a significant percentage of the NW European wintering population.

What they eat: Seeds and small invertebrates.

UK conservation status[9]: Amber

UK Wintering (Individual): 435,000[10] **UK Breeding (Pairs):** (2,700-4,750)[11]

Importance level: 87% of biogeographic population (NW Europe) winters in UK
1.0 – 1.9% of the biogeographic population breeds in UK

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6 Eaton et al (2015). See footnote 1
7 Woodward et al (2020). See footnote 2
8 Woodward et al (2020). See footnote 2
9 Eaton et al (2015). See footnote 1
10 Woodward et al (2020). See footnote 2
11 Woodward et al (2020). See footnote 2

1.5 Nightjar (*Caprimulgus europaeus*)



Key Information: Nightjars are nocturnal birds and can be seen hawking for food at dusk and dawn. With pointed wings and a long tails their shape is similar to a kestrel or cuckoo. Their cryptic, grey-brown, mottled, streaked and barred plumage provides ideal camouflage in the daytime. They have an almost supernatural reputation with their silent flight and their mythical ability to steal milk from goats. The first indication that a nightjar is near is usually the male's churring song, rising and falling with a ventriloquial quality.

What they eat: Insects - moths and beetles.

UK conservation status[12]: Amber

UK Breeding (Males): 4,600 (3,700-5,500)[13]

Importance level: 0.3 – 0.9% of the international population breeds in UK

1.6 Stone-curlew (*Burhinus oedicnemus*)



Key Information: A strange, rare summer visitor to southern England and East Anglia, the stone-curlew is a crow-sized bird with a large head, long yellow legs and relatively long wings and tail. Active at night, its large yellow eyes enable it to locate food when it is dark. It is not related to curlews and gets its name from its curlew-like call. It is listed under Schedule 1 of the Wildlife and Countryside Act.

What they eat: Invertebrates that are found on the ground.

UK conservation status[14]: Amber

UK Breeding (Pairs): 365^{+15}

Importance level: 0.12 - 0.2% of the international population breeds in UK

[12] Eaton et al (2015). See footnote 1
[13] Woodward et al (2020). See footnote 2
[14] Eaton et al (2015). See footnote 1
[15] Woodward et al (2020). See footnote 2

1.7 Avocet (*Recurvirostra avosetta*)

Key Information: The avocet is a distinctively-patterned black and white wader with a long up-curved beak. This Schedule 1 species is the emblem of the RSPB and symbolises the bird protection movement in the UK more than any other species. Its return in the 1940s and subsequent increase in numbers represents one of the most successful conservation and protection projects.



What they eat: Aquatic insects and their larvae, crustaceans and worms.

UK conservation status[16]: Amber

UK Wintering (Individual): 8,700[17] **UK Breeding (Pairs):** 1,950[18]

Importance level: 1.85 - 3.12% of biogeographic population winters in UK
0.83 – 1.39% of the international population breeds in UK

1.8 Sandwich tern (*Sterna sandvicensis*)

Key Information: The Sandwich tern is a very white tern, with a black cap on its head, a long black bill with a yellow tip and short black legs. In flight it shows grey wedges on its wings tips and it has a short forked tail. In the UK, many of the important colonies survive because they are on nature reserves.



What they eat: Fish such as sandeels, sprats and whiting.



UK conservation status[19]: Amber

UK Breeding (Pairs): 14,000 (13,000-15,000)[20]

Importance level: 14.0 – 18.8% of the biogeographic population (W Europe/W Africa) breeds in the UK

[16] Eaton et al (2015). See footnote 1
[17] Woodward et al (2020). See footnote 2
[18] Woodward et al (2020). See footnote 2
[19] Eaton et al (2015). See footnote 1
[20] Woodward et al (2020). See footnote 2

1.9 Little tern (*Sternula albifrons*)



Key Information: This delightful chattering seabird is the UK's smallest tern. It is short-tailed and has a fast flight. Its bill is a distinctive yellow with a black tip. It is noisy at its breeding colony where courtship starts with an aerial display involving the male calling and carrying a fish to attract a mate, which chases him up high before he descends, gliding with wings in a 'V'. It is listed as a Schedule 1 species in The Wildlife and Countryside Act.

What they eat: Fish.

UK conservation status[21]: Amber

UK Breeding (Pairs): 1,450[22]

Importance level: 11.6 – 15.3% of the biogeographic population (Europe north of Mediterranean) breeds in the UK

1.10 Common tern (*Sterna hirundo*)



Key Information: These delightful silvery-grey and white birds have long tails which have earned them the nickname 'sea-swallow'. They have a buoyant, graceful flight and frequently hover over water before plunging down for a fish. They are often noisy in company and breed in colonies. The common tern is the tern species most likely to be found inland.

What they eat: Fish.

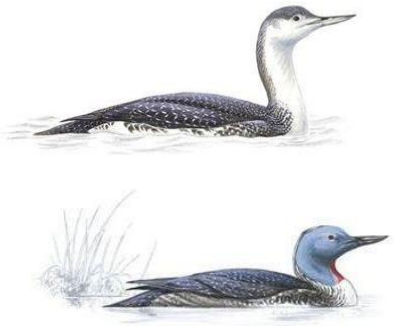
UK conservation status[23]: Amber

UK Breeding (Pairs): 11,000 (8,900-13,500)[24]

Importance level: 8.1 – 15.9% of the biogeographic population (S & W Europe) breeds in the UK

[21] Eaton et al (2015). See footnote 1
[22] Woodward et al (2020). See footnote 2
[23] Eaton et al (2015). See footnote 1
[24] Woodward et al (2020). See footnote 2

1.11 Red-throated diver (*Gavia stellata*)



Key Information: The smallest of the UK's divers, its grey-brown plumage and up-tilted bill readily distinguish it from the other species. In summer it has a distinctive red throat. They usually jump up to dive and can stay underwater for a minute and a half. They are very ungainly on land, only coming ashore to breed. They are listed as a Schedule 1 species under The Wildlife and Countryside Act.

What they eat: Fish.

UK conservation status[25]: Green

UK Wintering (Individual): 21,500[26] **UK Breeding (Pairs):** 1,250 (1,000-1,550)[27]

Importance level: 7.2% of biogeographic population (NW Europe) winters in UK
0.5 - 1.4% of the biogeographic population breeds in UK

1.12 Bittern (*Botaurus stellaris*)



Key Information: The bittern is a thickset heron with all-over bright, pale, buffy-brown plumage covered with dark streaks and bars. It flies on broad, rounded, bowed wings. A secretive bird, very difficult to see, as it moves silently through reeds at water's edge, looking for fish. The males make a remarkable far-carrying, booming sound in spring. It's very small, reedbed-dependent population make it an Amber List species. It is also a Schedule 1 species in the Wildlife and Countryside Act.

What they eat: Fish, amphibians and insects.

UK conservation status[28]: Amber

UK Wintering (Individual): 795[29] **UK Breeding (Males):** 191[+30]

Importance level: 2.1% of the international population breeds in UK
8.7% of the international population winters in UK

[25] Eaton et al (2015). See footnote 1
[26] Woodward et al (2020). See footnote 2
[27] Woodward et al (2020). See footnote 2
[28] Eaton et al (2015). See footnote 1
[29] Woodward et al (2020). See footnote 2
[30] Woodward et al (2020). See footnote 2

1.13 Marsh harrier (*Circus aeruginosus*)





Key Information: The largest of the harriers, the marsh harrier can be recognised by its long tail and light flight with wings held in a shallow 'V'. It is distinguishable from other harriers by its larger size, heavier build, broader wings and absence of white on the rump. Females are larger than males and have obvious creamy heads. Marsh harriers are a Schedule 1 listed bird on The Wildlife and Countryside Act.

What they eat: Small birds and mammals.

UK conservation status[31]: Amber

UK Breeding (Pairs): (590-695)[32]

Importance level: 0.3 – 0.7% of the international population breeds in UK

1.14 Hen harrier (Circus cyaneus)





Key Information: Of the UK's birds of prey, this is the most intensively persecuted. Once predating free-range fowl, earning its present name, its effect on the number of grouse available to shoot is the cause of modern conflict and threatens its survival in some parts of the UK, particularly on the driven grouse moors of England and Scotland.

While males are a pale grey colour, females and immatures are brown with a white rump and a long, barred tail which give them the name 'ringtail'. They fly with wings held in a shallow 'V', gliding low in search of food, which mainly consists of meadow pipits and voles. The Orkney population is famous for being polygynous, with males sometimes mating with multiple females on the island. They are listed as a Schedule 1 species under The Wildlife and Countryside Act.

What they eat: Small birds and mammals.

UK conservation status[33]: Red

UK Breeding (Pairs): 545[34]

Importance level: 1.0 – 1.8% of international population breeds in UK

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[31] Eaton et al (2015). See footnote 1
[32] Woodward et al (2020). See footnote 2
[33] Eaton et al (2015). See footnote 1
[34] Woodward et al (2020). See footnote 2

1.15 Bearded tit (*Panurus biarmicus*)





Key Information: The bearded tit is a Schedule 1 species in the Wildlife and Countryside Act. They are brown, long-tailed birds, usually seen flying rapidly across the top of a reedbed. Males have black 'moustaches' rather than 'beards'. They are sociable and noisy, their 'ping' calls often being the first clue to their presence.

What they eat Insects, insect larvae, spiders and seeds.

UK conservation status[35]: Green

UK Breeding (Pairs): 695[36]

Importance level: 0.2 – 0.3% of the European population breeds in the UK

1.16 Woodlark (*Lullula arborea*)



Key Information: The woodlark is a streaky brown bird, with a buffy-white eye-stripe which meets across the nape. It has a well-developed crest on its crown which is not always conspicuous. In flight the peculiarly short tail and broad, rounded wings are noticeable and the deeply undulating flight with closed wing glides is characteristic. Some UK breeding birds spend the winter on the continent. Woodlarks are listed under Schedule 1 of The Wildlife and Countryside Act.

What they eat Seeds and insects.

UK conservation status[37]: Green

UK Breeding (Pairs): 2,300 (1,850-2,750)[38]

Importance level: 0.05 – 0.1% of the European population breeds in the UK

[35] Eaton et al (2015). See footnote 1
[36] Woodward et al (2020). See footnote 2
[37] Eaton et al (2015). See footnote 1
[38] Woodward et al (2020). See footnote 2

1.17 Dartford warbler (*Sylvia undata*)



Key Information: This small, dark, long-tailed warbler is resident in the UK and has suffered in the past from severe winters. The Dartford warbler's population crashed to a few pairs in the 1960s, since when it has gradually recovered, increasing in both numbers and range. It will perch on top of a gorse stem to sing, but is often seen as a small flying shape bobbing between bushes.

What they eat Insects and other invertebrates.

UK conservation status[39]: Amber

UK Breeding (Pairs): 2,200[40]

Importance level: 0.2 – 0.4% of the international population breeds in the UK

[39] Eaton et al (2015). See footnote 1

[40] Woodward et al (2020). See footnote 2

Appendix 2
to
Written Submission

for the
Royal Society for the Protection of Birds
and Suffolk Wildlife Trust

Submitted for Deadline 2
2 June 2021

Planning Act 2008 (as amended)

In the matter of:

Application by NNB Generation Company (SZC) Limited for an Order Granting Development Consent for

The Sizewell C Project

Planning Inspectorate Ref: EN010012
RSPB Registration Identification Ref: 20026628
Suffolk Wildlife Trust Registration Identification Ref: 20026359

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Appendix 1: Species Accounts

Appendix 2: Footprint Ecology reports regarding recreational impacts:

- **Liley, D. & Saunders, P. (2020a) Impacts of recreation related to Sizewell C and implications for European sites. Unpublished report by Footprint Ecology**
- **Liley, D. & Saunders, P. (2020b) Review of Sizewell C application documents and evidence in relation to recreation impacts. Unpublished report by Footprint Ecology.**

Appendix 3: Natterjack toad report:

- Amphibian and Reptile Conservation (2020). Sizewell C Project: comments on natterjack toad mitigation proposals. Report to Suffolk Wildlife Trust. ARC report. ARC, Bournemouth.

Appendix 4: List of References

FOOTPRINT ECOLOGY



Impacts of recreation related to Sizewell C and implications for European sites

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Summary

This report has been commissioned by the National Trust and the RSPB in light of concerns regarding proposals for Sizewell C Power Station and potential nature conservation impacts from recreation use. The area around Sizewell is of outstanding importance for nature conservation and contains a mix of habitats and species that are unique in the UK. The area is also popular for recreation, which includes tourists, day visitors, and local residents. There is a difficult balancing act to ensure access and conservation do not conflict in a relatively small area.

The Sizewell C proposal would result in changes in recreation use as it would involve a large construction workforce that would be living and working in the area. In addition, construction work would displace existing recreation use (for example through footpath diversions, noise, traffic, etc). We review the impacts of recreation on the nature conservation interest of relevant sites, focussing on the European sites (i.e. those that are subject to particularly strict protection) and their qualifying features.

We highlight a wide range of vulnerable features and risks to their conservation interest resulting from recreation use. Likely significant effects are clear for a range of sites and interest features and, in particular, we highlight:

- Annual vegetation of drift lines (Minsmere to Walberswick Heaths & Marshes SAC), risks from damage and contamination;
- Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC), risks from damage and contamination;
- European dry heaths (Minsmere to Walberswick Heaths & Marshes SAC), risks from fire and contamination;
- Wintering waterbirds, including White-fronted Goose (Minsmere to Walberswick SPA), risks from disturbance;
- Breeding birds, including Nightjar, Avocet, and Little Tern (Minsmere to Walberswick SPA), risks from disturbance;
- Breeding Nightjar (Minsmere to Walberswick SPA), risks from fire, and;
- Breeding Woodlark and Nightjar (Sandlings SPA), risks from disturbance and fire.

People coming to the area to live and work in the vicinity will seek nearby greenspace for recreational use that is not necessarily related or connected to the area's conservation interest, and such use will include exercise and dog walking. It may extend to sea angling, barbeques, swimming, and beach recreation. Such use is likely to be away from areas perceived as 'nature reserves' (i.e. where there are hides, wardens, and a fee to enter) and instead focus their use at more peripheral locations. These are often still part of the European site and do not have necessary infrastructure to manage access. We would expect

construction workers based in the area to utilise more informal parking and locations where they can easily access beaches, quiet areas of heath, etc. Use by construction workers is also likely to take place in the early mornings, evenings, and around shift work and therefore not necessarily fit with peak visitor use and the current visitor profile.

We provide an overview of the mitigation that is likely to be necessary, highlighting that a package of measures will be required, and this will need to include monitoring to help pick up emerging issues.

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This report has been commissioned by the National Trust to provide background and help inform responses by the National Trust and the RSPB to the proposals for a new Sizewell C Power Station. Our thanks to Emma Hay (National Trust) for commissioning the work and in addition to Jacqui Miller (RSPB) and Adam Rowlands (RSPB) for useful discussion and context.

1. Introduction

Overview

1.1 The RSPB and National Trust are concerned about the potential for displacement of recreational users from the vicinity of the Sizewell C development (SZC) to neighbouring designated sites during the construction period.

1.2 Risks from recreation potentially relate to:

- Use of nearby sites for recreation by construction workers, including those potentially housed in purpose-built accommodation at Eastbridge, and;
- Displacement of visitors who would otherwise have gone to Sizewell, to access the beach and coastline around the power station and have been displaced due to the traffic, construction work and changes to access.

1.3 Displacement of visitors from Sizewell is likely to primarily relate to those using the large public car-park just to the south of Sizewell A, who might be expected to shift to other locations. There may also be some displacement in visitor routes around the general area, for example visitors at Dunwich and Minsmere may be deterred from walking as far south as they might have done otherwise due to the construction work.

1.4 The National Trust, with the RSPB, have therefore commissioned this report to review the sensitivity of designated site features to recreation impacts and to consider the potential for suitable mitigation.

2. Relevant European sites

2.1 For recreation issues associated with new housing in Suffolk, the Recreation Avoidance Mitigation Strategy (Hoskin, Liley, & Panter, 2019) highlights how different mitigation strategies around the country define zones of influence (drawing on visitor data to show where changes in housing are likely to result in changes in access use) ranging from 5-15km. Drawing on data from the Suffolk Sandlings (Cruickshanks, Liley, & Hoskin, 2010) and the Deben Estuary (Lake, et al., 2014), the Suffolk strategy uses 13km. This is helpful as a broad guide to indicate where impacts may be expected.

2.2 Based on this guide, our knowledge of the nearby European sites and their draw for recreation and their accessibility from Sizewell/Eastbridge (taking into account the road network), we would anticipate the potential for risks at the following sites:

- Alde, Ore & Butley Estuaries SAC;
- Benacre to Easton Bavents Lagoons SAC;
- Orfordness - Shingle Street SAC;
- Minsmere to Walberswick Heaths & Marshes SAC;
- Alde-Ore Estuary SPA;
- Benacre to Easton Bavents SPA;
- Minsmere to Walberswick SPA;
- Sandlings SPA;
- Alde - Ore Estuary Ramsar, and;
- Minsmere - Walberswick Ramsar.

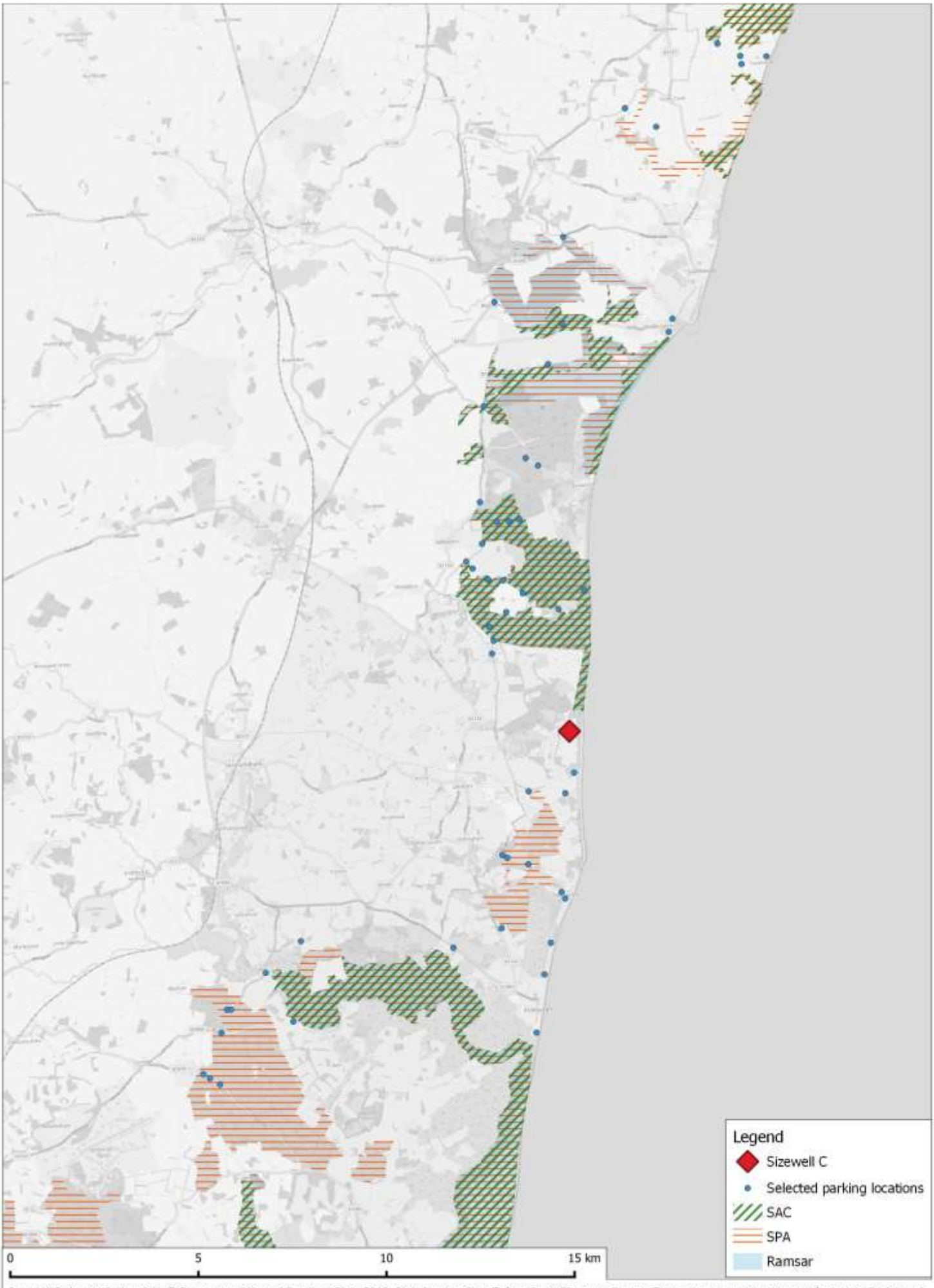
2.3 These European sites are shown in Map 1, where a selection of parking locations are also shown. These parking locations do not represent a comprehensive review of all potential parking locations and have not been systematically mapped from site visits. However, they are drawn on our knowledge of the sites and reflect locations where access might be expected to be deflected/change. There are a wide range of possible parking locations not mapped (see for example Cruickshanks et al., 2010) and those shown are intended as a guide only, providing a broad indication of the key areas where access might change.

2.4 We have discounted the following European sites as not relevant, even though they are potentially within the kinds of distance whereby displacement might be expected:

- Southern North Sea SAC, as this is a marine site designated for Harbour Porpoise *Phocoena phocoena*;

- Outer Thames Estuary SPA. as this is a marine site classified for foraging terns and non-breeding Red-throated Diver *Gavia stellata*;
- Staverton Park & the Thicks SAC, as this site is linearly c.15km from Sizewell, with travel distance larger still due to the barrier of the Alde-Ore Estuary. Public access at the SAC is limited to a public footpath close to the boundary of the SAC, and;
- Dew's Ponds SAC, as this is an inland site, comprising various ponds on arable land, and designated for Great-crested Newts *Triturus cristatus*. It is unlikely to draw visitors away from the coast.

Map 1: Overview of sites and parking provision



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3. Impacts from recreation

3.1 The nature conservation impacts of recreation are summarised in a number of general reviews (e.g. Buckley, 2004; Liddle, 1997; Liley et al., 2010; Lowen, Liley, Underhill-Day, & Whitehouse, 2008).

3.2 We group impacts from recreation under the following broad headings, which we use to structure this section:

- Disturbance;
- Damage;
- Fire;
- Contamination, and;
- Impacts on site management.

Disturbance

Overview

3.3 Disturbance occurs where human activity influences an animal's behaviour or survival. By far the majority of the literature (and there are thousands of studies), focuses on birds (Brawn, Robinson, & Thompson III, 2001; Hill et al., 1997; for general reviews see Hockin et al., 1992; Lowen et al., 2008; Showler, 2010; Steven, Pickering, & Castley, 2011; Whitfield, Ruddock, & Bullman, 2008). Disturbance can also affect mammals, herptiles (see Edgar, 2002 for review) and invertebrates.

General principles

3.4 The presence of people in the countryside will influence wildlife in many ways. For many species, people or their pets (e.g. dogs) are a potential threat and as such it is to be expected that the response will be to modify behaviour, for example fleeing. The relative trade-off as to when to change behaviour and respond to the threat will relate to the perceived scale of the threat and the costs involved (e.g. lost foraging time). This perspective can be used to understand the behavioural responses to people and led one author to describe human disturbance as predation-free predators (Beale & Monaghan, 2004).

3.5 With people (and their pets) viewed as potential predators, there is clearly a greater threat posed (and therefore a greater behavioural response) when, for example, there are more people, in larger groups (Beale & Monaghan,

2004, 2005) or when people approach directly (Smith-Castro & Rodewald, 2010) or faster (Bellefleur, Lee, & Ronconi, 2009).

3.6 The presence of people may also draw particular predators, for example a study in America showed the crow (corvid) populations were centred around campgrounds (Marzluff & Neatherlin, 2006) while Kays *et al.* (2017) used camera traps to show a range of predators actively selected human-made paths. As such the presence of people may also influence the distribution and abundance of predators with a knock-on effect for potential prey species.

Impacts

3.7 Disturbance can therefore have a range of different impacts, potentially affecting distribution, breeding success, and health. Impacts can be chronic, for example otherwise suitable nesting habitat being completely avoided (e.g. Liley & Sutherland, 2007) or more short-term in nature, for example birds becoming alert and then resuming the initial activity (e.g. Fernandez-Juricic, Jimenez, & Lucas, 2001). Birds might be temporarily displaced from particular locations and such behavioural responses will have some energetic costs, even if only very short term in duration. Impacts can also include indirect mortality, for example through increased predation associated with disturbance (e.g. Brambilla, Rubolini, & Guidali, 2004).

3.8 There are also examples of direct predation by pet dogs, for example dogs were recorded as predators of nests and incubating adult Ringed Plovers *Charadrius hiaticula* on Lindisfarne (Pienkowski, 1984). Some studies have shown evidence of accidental trampling of nests and young, including herptiles (Edgar, 2002) and birds (Liley & Sutherland, 2007). There are also studies that show direct trampling impacts by people of invertebrates (Ciach, Maślanka, Krzus, & Wojas, 2017).

3.9 Much harder to measure and record are physiological effects, for example related to stress, and these may in turn affect fitness. While studies are limited, there is evidence of physiological effects in terms of increased heart rate (Ellenberg, Mattern, & Seddon, 2013) and stress-hormones (Thiel, et al,. 2011).

Relevant qualifying features

3.10 At coastal sites, there are a range of studies that show reduced breeding success for breeding birds such as gulls and terns (Medeiros et al., 2007;

Robert & Ralph, 1975; Sandvik & Barrett, 2001). Little Terns *Sternula albifrons* are particularly vulnerable to disturbance as they nest on open sandy/shingle beaches where access is often concentrated. They will avoid nesting in otherwise suitable habitat where there are high disturbance levels (i.e. more people) (Ratcliffe, et al., 2008). Colonies will vary in size and location between years and there is likely to be interchange between different sites along the Suffolk coast over time.

3.11 Disturbance to wintering and passage waterbirds can result in:

- A reduction in the time spent feeding due to repeated flushing/increased vigilance (Bright, et al, 2003; Fitzpatrick & Bouchez, 1998; Stillman & Goss-Custard, 2002; Thomas, Kvitek, & Bretz, 2003; Yasué, 2005);
- Increased energetic costs (Nolet, et al, 2002; Stock & Hofeditz, 1997);
- Avoidance of areas of otherwise suitable habitat, potentially using poorer quality feeding/roosting sites instead (Burton, et al., 2002; Burton, Rehfisch, & Clark, 2002; Cryer, et al.1987; Gill, 1996), and;
- Increased stress (Payne et al., 2012; Sharpe, et al., 2009).

3.12 Disturbance arising from human activity has been shown to negatively affect the foraging efficiency and fat accumulation rate of diving ducks (Knapton, Petrie, & Herring, 2000), with pedestrian and cycle-based activity shown to have a greater impact in the winter than vehicular disturbance (Pease, Rose, & Butler, 2005).

3.13 For species associated with reedbeds, there is limited evidence of disturbance impacts for Bitterns (Underhill-Day, 1981), but while recreational disturbance of breeding birds at the nest is less likely, it has been suggested that human activity on the edges of reed-beds and on waterway margins may have an impact (Underhill-Day & Wilson, 1978). There is evidence of disturbance effecting breeding success for Marsh Harriers, for example. Underhill-Day (1984) found that 9% of nests were deserted as a result of human disturbance, and a range of activities were involved that included photography, walking, and fishing. Other studies have highlighted increased energetic costs imposed upon nesting Marsh Harriers by disturbance (Fernandez & Azkona, 1993).

3.14 For heathland bird species there are a range of disturbance studies. For both Nightjar and Woodlark, studies have shown recreation use affects the distribution of birds within sites, such that busy areas are avoided (Liley et al. 2006; Mallord, et al. 2007; Lowe, Rogers & Durrant, 2014). For Nightjars there

is also evidence of breeding success being lower on busier sites and busier parts of sites (Murison 2002). For Woodlarks at least, there are clear population-level impacts as a result of the presence of people on the heaths (Mallord et al. 2007).

3.15 Little work has been undertaken on disturbance impacts for wintering Hen Harrier. Traditional roost sites are identified as places where access management measures, or exclusions/restrictions, should be implemented in relation to CRoW (Brown & Langston, 2001). A well-known hen harrier roost site in Dorset was abandoned in 1997, with local counters believing that increased access by walkers and other recreational activities was the cause (Lowen et al., 2008).

3.16 Impacts for different European site qualifying features are summarised in Table 1.

Table 1: European site qualifying features identified as being susceptible to disturbance impacts from recreational activity.

Qualifying feature	European site(s)	Seasonality	References	Notes
Breeding wildfowl (i.e. Gadwall, Shoveler, and Teal)	Minsmere-Walberswick SPA; Minsmere-Walberswick Ramsar	Spring and Summer		Potential for disturbance at wetland breeding localities if walkers/dogs are able to access water's edge.
Wintering wildfowl (i.e. Gadwall, Shoveler, and White-fronted Goose)	Minsmere-Walberswick SPA	Winter		Recreational disturbance could potentially influence where birds feed, time spent feeding and flushing will have energetic consequences.
Marsh Harrier	Minsmere-Walberswick SPA; Alde-Ore Estuary SPA; Benacre to Eastern Bavents SPA; Minsmere-Walberswick Ramsar	Spring and Summer	(Fernandez & Azkona, 1993; Underhill-Day, 1984)	Both males and female spend less time near the nest, bring in fewer food items, and exhibit stress-related behaviours when disturbed.
Hen Harrier	Minsmere-Walberswick SPA	Winter	(A. Brown & Langston, 2001; Lowen et al., 2008)	Winter roost sites are likely to be susceptible to disturbance.
Bittern	Minsmere-Walberswick SPA; Benacre to Eastern Bavents SPA; Minsmere-Walberswick Ramsar	Spring and Summer	(Underhill-Day & Wilson, 1978)	Impacts more relevant to feeding birds and assumed access difficult and unlikely to nest sites
Bearded Tit	Minsmere-Walberswick SPA; Minsmere-Walberswick Ramsar	Spring and Summer		Potential scope for disturbance if accessing extensive reedbed localities.
Avocet	Minsmere-Walberswick SPA; Alde-Ore Estuary SPA; Alde-Ore Ramsar; Minsmere-Walberswick Ramsar	Spring and Summer	(Scarton, 2018)	Any access to breeding localities could lead to disturbance/abandonment. Birds respond at quite considerable distances to presence of people.
Avocet	Alde-Ore Estuary SPA	Winter		
Lesser Black-backed Gull	Alde-Ore Estuary SPA; Alde-Ore Ramsar	Spring and Summer		Potential for disturbance effects at breeding colonies, although recreational access considered unlikely on the Alde-Ore Estuary due to isolated nature of colonies.
Sandwich Tern	Alde-Ore Estuary SPA	Spring and Summer	(Brown & Grice, 2005; Garthe &	Potential for abandonment, damage to eggs/chicks, and increased levels of natural predation at colonies. Nests

Qualifying feature	European site(s)	Seasonality	References	Notes
			Flore, 2007; Lloyd, Bibby, & Everett, 1975)	in large colonies and single access events can cause whole colonies to desert.
Little Tern	Minsmere-Walberswick SPA; Alde-Ore Estuary SPA; Benacre to Eastern Bavents SPA	Spring and Summer	(Medeiros et al., 2007; Ratcliffe, Schmitt, Mayo, Tratalos, & Drewitt, 2008)	Decreased productivity, and increased levels of natural predation, arising from recreational activity. Birds also avoid nesting on beaches with high levels of access. Colonies are mobile and can shift between years, responding to habitat change and other factors.
European Nightjar	Minsmere-Walberswick SPA; Sandlings SPA	Spring and Summer (to late August)	(Liley, Clarke, Mallord, & Bullock, 2006; Lowe, Rogers, & Durrant, 2014; Murison, 2002)	Recreational activity can lead to avoidance of certain areas within a site, and also negatively impact productivity. Dunwich Forest population is likely to be functionally linked to birds within the Minsmere-Walberswick SPA and Sandlings SPA.
Woodlark	Sandlings SPA	All year (although increased impact in Spring and Summer). Potentially particularly vulnerable when settling on territories in Feb/March.	(Eyre & Baldwin, 2014; Mallord, Dolman, Brown, & Sutherland, 2007)	Recreational activity can lead to displacement or avoidance of certain areas within a site, and also negatively impact productivity. Studies with nest cameras have shown some instances where nests lost due to joggers or bikes crushing nests. We consider birds within the Minsmere-Walberswick SPA and within Dunwich Forest population to be functionally-linked to the Sandlings population.
Redshank	Alde-Ore Estuary SPA; Alde-Ore Ramsar	Winter	(Fitzpatrick & Bouchez, 1998)	Birds prone to flushing and lost feeding time following recreational disturbance
Ruff	Alde-Ore Estuary SPA	Winter		Wintering birds feeding on areas of wet grassland may be susceptible to disturbance effects.

Damage

Overview

3.17 Recreational activities can lead to changes in soil characteristics and ultimately lead to erosion. Although erosion brought about by recreational activities is small compared to natural factors it can none the less an important form of soil degradation (Holden et al., 2007). Changes to substrates can in turn lead to changes in the ecological communities they support.

3.18 At lower levels of use, the main impact is on vegetation and is largely mechanical (Bayfield & Aitken, 1992; Liddle, 1997) while higher levels of use will also affect substrates. Light use may cause a slight decrease in vegetation cover, and a decline in the incidence of flowering. Bare ground may be colonised by trampling resistant species. Heavier ground pressure leads to greater losses of vegetation. Significant erosion can be expected where the plant cover falls below 70% (Liddle, 1997), but erosion can commence before this level is reached (Kuss & Morgan, 1984). As loss of vegetation takes place, there is disruption and progressive loss of soil horizons by direct physical abrasion or loosening and indirectly by water and wind erosion. Important changes in soil structure and chemistry can result from compaction. Poor permeability to water can increase surface run-off, and reduced aeration can result in anaerobic conditions and poor root growth.

3.19 Trampling has been shown to alter the amount of litter present (Bayfield & Brookes, 1979), soil water content, soil temperature and chemistry (Liddle, 1997). There is also some evidence that trampling and resulting compaction can effect soil health and the mycorrhizae or bacteria present (with knock-on implications for plant health) (Fay, 2014).

3.20 Different recreational activities can have a significantly different impact. In general, walking is likely to be less damaging that horse riding, cycling, or motorised vehicles. For example, Weaver and Dale (1978) showed that horses were substantially more damaging, and motorcycles slightly more damaging, than hikers in grassland and woodland in the US Pacific Northwest. Thurston and Reader (2001) suggest that mountain bikes cause the same amount of damage as hikers in deciduous woodland, although MacIntyre (1991) and Rees (1990) show that mountain bikes may cause slightly more damage than foot traffic depending on the type of habitat.

Relevant qualifying features

3.21 Heather-dominated vegetation is very susceptible to trampling damage, though there may be some differences related to individual species response and soil conditions. In summer and winter trials on undamaged lowland heathland in England (Harrison, 1981), it was shown that 400 passes in the first summer of the experiment, caused heather cover to fall to about 50%, and by 800 passes it was less than 10%. The vegetation failed to recover in the period following the experimental trampling, after winter only, summer only, or all season trampling.

3.22 Seasonal and habitat response was tested in trials on heathland in Brittany (Gallet & Roze, 2001a) and though there were some differences, in all cases trampling led to a great decrease in vegetation cover, with the vegetation cover varying between 0 and 50% under 750 passes. Dry heathland was more resistant than mesophilous (humid) heath and significantly so with winter trampling, but both heath types were equally vulnerable in wet conditions. Gorse was more resilient than heathers; and younger dwarf shrubs were less vulnerable than older plants.

3.23 Though trampling can damage the dwarf shrub community of heathland, there are some aspects of the habitat that need the canopy to be broken, even to the extent of bare ground being sustained. Bare ground and early successional habitats are a very important component of the heathland ecosystem, important for a suite of plants, invertebrates and reptiles (Byfield & Pearman, 1996; Key, 2000; Lake & Underhill-Day, 1999). Typically, small, low-growing, herbs with low competitive capacity require these open conditions and lack of suppression by a taller canopy. Some may be ruderals or annuals that can only survive in such conditions.

3.24 Some kind of physical disturbance is usually required to create these bare ground habitats, and hence a certain level of physical disturbance, including erosion resulting from trampling, can be beneficial. However, the level of disturbance required is difficult to define and is likely to vary between sites (Lake, Bullock & Hartley 2001). There are likely to be optimum levels of use that maintain the bare ground habitats but do not continually disturb the substrate. Such levels of use have never been quantified, nor is it known whether sporadic use is likely to be better at maintaining bare ground habitats than low level, continuous use.

3.25 Research has shown that shingle vegetation, strandline communities, and reedbed are easily damaged by trampling, and that this can lead to a decrease in plant species diversity within shingle communities particularly (Penny Anderson Associates, 2009). This has been shown to be the case historically on the Orford Shingle (Fuller & Randall, 1988). Trampling in reedbeds is likely to be constrained by high water levels for much of the year, although may prove an issue during periods of low water and/or at the periphery of the locality. Such peripheral trampling/poaching is also of potential import for coastal lagoons and other isolated waterbodies.

3.26 Fen habitats are also highly susceptible to trampling damage, although the high water table present in tall herb fen may dissuade access away from the periphery. Nevertheless. Due to the wet, or seasonally wet, ground present within fens, the habitat is extremely susceptible to trampling damage where access is possible (Penny Anderson Associates, 2009). Experimental trampling carried out over several years within fens in Norway led to severe damage (Arneson, 1999) and results from Scotland showed that taller fen grasses were less resilient to trampling than sedges (Rees & Tivy, 1977).

3.27 Impacts of recreation upon saltmarsh have been relatively little studied, due to the often inaccessible nature of the terrain (Penny Anderson Associates, 2009), although several studies identify trampling effects (Hewitt, 1973; Schofield, 1967). The creation of distinct pathways across saltmarsh, potentially leading to changes in vegetation community structure/composition, has also been noted (Chandrasekara & Frid, 1996). In Coombes (2007) review of trampling effects on different East Anglian coastal habitats, the data show saltmarshes as more robust compared to certain other coastal habitats.

3.28 Impacts upon notable plant and invertebrate species within the qualifying habitats is likely to vary by habitat type. Some light trampling within more robust fen habitats and reedbed may prove beneficial for invertebrates especially, although notable plants/invertebrates associated with sand dune and coastal shingle are particularly susceptible to damage via trampling (Penny Anderson Associates, 2009).

3.29 Impacts for different European site qualifying features are summarised in Table 2.

Table 2: European site qualifying features identified as being susceptible to physical damage arising from recreational activity

Qualifying feature	European site(s)	Seasonality	References	Notes
H4030 European dry heaths	Minsmere-Walberswick Heaths & Marshes SAC	During the growing season. Lichens more vulnerable during the summer when dry.	(Byfield & Pearman, 1996; Gallet & Roze, 2001b; Harrison, 1981; Key, 2000; Lake et al., 2001; Lake & Underhill-Day, 1999; Penny Anderson Associates, 2009)	Some limited trampling/disturbance required to maintain important bare ground habitat components.
H1210 Annual vegetation of drift lines	Minsmere-Walberswick Heaths & Marshes SAC; Orfordness-Shingle Street SAC	During growing season.	(Doody & Randall, 2003)	Dynamic habitat type, susceptible to impact of tidal /storm action. Pioneer species susceptible to trampling impacts.
H1220 Perennial vegetation of stony banks	Minsmere-Walberswick Heaths & Marshes SAC; Orfordness-Shingle Street SAC	During the growing season. Lichens more vulnerable during the summer when dry.	(Doody & Randall, 2003; Lowen et al., 2008)	Highly susceptible to trampling effects, with often isolated/patchy nature of vegetation on shingle potentially exacerbating impact more so than in more contiguous habitat types, and making recovery and recolonization more difficult/slower. Areas with lichen cover particularly vulnerable.
H1150 Coastal lagoons	Benacre to Easton Bavents Lagoons SAC; Orfordness-Shingle Street SAC	Throughout year	(Lowen et al., 2008; Penny Anderson Associates, 2009; Saunders, Selwyn, Richardson, May, & Heeps, 2000)	Damage from pedestrian access likely to be limited to periphery of habitat, where shallow water may support specialised communities. Entry by people/dogs into deeper sections risks further physical damage to substrate. Trampling issues suggested as 'minimal' by Saunders *et al.*
H1330 Atlantic salt meadows	Alde-Ore & Butley Estuaries SAC	Throughout year	(Chandrasekara & Frid, 1996; Coombes, 2007;	Human access within saltmarsh habitats constrained by topography and tidal action, forcing creation of distinct,

Qualifying feature	European site(s)	Seasonality	References	Notes
			Hewitt, 1973; Lowen et al., 2008; Penny Anderson Associates, 2009; Schofield, 1967)	and sometimes heavily utilised, pathways within accessible locations. Trampling has been shown to cause physical damage to habitat and changes in community structure. Saltmarsh habitats potentially less vulnerable compared to other coastal habitats.
H1130 Estuaries	Alde-Ore & Butley Estuaries SAC	Throughout year	(Lowen et al., 2008; Penny Anderson Associates, 2009)	Dynamic/tidal nature of habitat means that lasting effects often concentrated in vicinity of high water mark. Possible for damage to estuarine substrate invertebrate and algal/pioneer plant communities via trampling. Many areas inaccessible or difficult to access.
H1140 Mudflats and sandflats	Alde-Ore & Butley Estuaries SAC	Throughout year	(Lowen et al., 2008; Penny Anderson Associates, 2009)	Dynamic/tidal nature of habitat means that lasting effects often concentrated in vicinity of high water mark. Possible for damage to substrate invertebrate and algal/pioneer plant communities via trampling but risks low given the difficulties with access..
Wetland habitats	Minsmere-Walberswick Ramsar	Throughout year	(Arneson, 1999; Lowen et al., 2008; Penny Anderson Associates, 2009; Rees & Tivy, 1977)	Physical damage likely to be limited to periphery during periods of high water. Drier parts of reedbeds can be readily accessible and trampling risks possible. Would require relatively high or repeated incursions for marked structural damage. Tall/waterlogged fen habitats highly susceptible to trampling impacts, including loss of less robust plant species leading to change in community structure. Potential for poaching of banks and trampling of marginal, and potentially emergent/submerged, vegetation, e.g. where dogs access water.
Nationally scarce plant species and British Red Data book invertebrates	Minsmere-Walberswick Ramsar	Throughout year		Impacts likely to relate to structural change to habitats, for which see above.

Fire

Overview

3.30 Fires in the countryside can be caused accidentally from discarded cigarettes, by sparks from a campfire, BBQs or from burning a dumped or stolen car, from fireworks, as a result of a controlled fire getting out of control, from discarded bottles in strong sunlight, from children playing with matches or similar, and from deliberate arson. Fire risks have increased with climate change (anon, 2017; Jolly et al., 2015; Moffat & Gazzard, 2019) and major fire incidents on nature reserves are becoming more common.

3.31 Based on 217 questionnaires from a sample of lowland heaths in Dorset, Kirby and Tantram (Tantram, Boobyer, & Kirby, 1999) found that 61% of fires were caused by arson, 8% from management fires getting out of control, 7% from bonfires and the remainder from camp fires, burning refuse, vehicle fires, property fire and sparks from a railway. The only natural cause of fire was from lightning. The same study noted that there was a widespread belief among the public and nature conservation professionals that most fires were deliberate and that children were often believed to be responsible (this would be most relevant on sites close to residential areas rather than remote uplands).

3.32 A number of studies have linked the incidence of fires with areas used by the public, or with the extent of urbanisation. In the Peak District National Park during 1970-1995, 84% of 324 recorded fires were next to roads, paths or within areas of open access, and many burnt areas on Exmoor are close to public roads (Miller & Miles, 1984). Kirby and Tantram (1999) noted that of the 26 lowland heathland SSSIs in Dorset with the highest number of fires, 1990-1998, 70% were located in or adjacent to urban areas, including the top nine.

Relevant qualifying features

3.33 Fires can have major impacts on the soil, vegetation, and fauna present, and recovery can take many years. After a fire where temperature and intensity are moderate, vegetation recovery will be largely influenced by the vegetation composition before the fire, although subsequent management, particularly grazing and trampling, will influence regeneration. The less palatable or better-adapted species may be favoured by grazing, so that, for example, on heathlands, cross leaved-heath and the more unpalatable

graminoids may benefit initially at the expense of heather. Pioneer communities, such as those on shingle banks, may be particularly impacted by fire, due to the small amounts of soil present and the time taken for those soils to develop (Randall, 2004). Any recovery following fire will be further hindered by the small size of the seedbank in pioneer communities.

3.34 A range of studies show impacts of fires for invertebrate populations. Recovery of the full community of unburnt areas can take as little as two years in grassland to 20 years in heathland habitats (Bell, Wheater, & Cullen, 2001; Panzer, 2002). While some species and communities can benefit from the open conditions following a fire or in regularly burned sites, others can be seriously depleted or even eliminated (Kirby, 2001). Fires is a frequently used tool in reedbed management, but if allowed to burn out of control it can damage invertebrate populations within reed stubble and litter, and potentially cause subsurface burns in areas with peat soils (Hawke & Jose, 1996).

3.35 Where fires are extensive, whole populations of invertebrates can be destroyed and large fires may cause local extinctions in less mobile species. Invertebrate groups which are most vulnerable to fire in open habitats are those present in the litter as eggs or larvae in spring when many fires take place, species with only one generation per annum and sedentary or flightless species or groups. These include molluscs, leafhoppers, grasshoppers and some butterfly and moth species (Kerney, 1999; Panzer, 2002).

3.36 Fire can also be particularly damaging to reptiles, and the occurrence of fires within heathland can also lead to habitat and/or nest destruction for important bird species (e.g. Nightjar and Woodlark).

3.37 Impacts for different European site qualifying features are summarised in Table 3.

Table 3: European site qualifying features identified as being susceptible to fires arising from recreational activity

Qualifying feature	European site(s)	Seasonality	References	Notes
H4030 European dry heaths	Minsmere-Walberswick Heaths & Marshes SAC	Can occur throughout year, particularly likely during prolonged hot/dry weather. Impact greater in spring/summer.	(Bell et al., 2001; Borghesio, 2009; Panzer, 2002; Underhill-Day, 2005)	Fire can be a useful management tool but such management is targeted to limited areas and only undertaken in the late winter. Wildfire can damage extensive areas of habitat and have devastating effects upon important bird, invertebrate, and plant species. Grazing levels can impact subsequent recovery, and long-term changes in habitat structure are possible if not carefully managed.
H1210 Annual vegetation of drift lines	Minsmere-Walberswick Heaths & Marshes SAC; Orfordness-Shingle Street SAC	Throughout year (potentially higher risk in summer months as increased access levels, use of disposable barbeques, etc)		Low risk due to patchy/tideline distribution and thin soils, impacts likely to be localised as fire unlikely to spread.
H1220 Perennial vegetation of stony banks	Minsmere-Walberswick Heaths & Marshes SAC; Orfordness-Shingle Street SAC	Throughout year (potentially higher risk in summer months as increased access levels, use of disposable barbeques, etc)	(Doody & Randall, 2003; Randall, 2004)	Higher risk than drift line communities due to more contiguous vegetation patches with more developed humus layer, but spread still unlikely and impacts possibly localised. Long-lasting impact due to time taken for soils to develop and paucity of seedbank available for recovery.
Wetland habitats	Minsmere-Walberswick Ramsar	Throughout year. Higher risk during periods of low water levels.	(Hawke & Jose, 1996)	Wildfire can destroy large areas of contiguous reedbed habitat and proved difficult to control. High water levels will preclude extensive damage to invertebrate communities within reed litter, but fires during periods of low water can lead to large die-offs of stubble and litter invertebrate populations.

Qualifying feature	European site(s)	Seasonality	References	Notes
Nationally scarce plant species and British Red Data book invertebrates	Minsmere-Walberswick Ramsar	Throughout year. Higher risk during periods of low water levels.	(Hawke & Jose, 1996)	Particular concern for drier parts of reedbeds, see above.

Contamination

Overview

3.38 Contamination risks include:

- Nutrient enrichment from dog fouling;
- Contamination of ponds and water bonds from incursions by dogs and people;
- Litter;
- Spreading of invasive species, and;
- Spreading of disease.

3.39 Air quality may also be affected via emissions from vehicles and fires, which may have resultant, diffuse, effects upon vegetation. Any changes in air quality resulting solely from increases in levels of recreational access will be difficult to isolate from changes as a result of general traffic, industry, nearby infrastructure, etc, and air quality effects are therefore not considered further within this report.

3.40 A relatively limited number of studies have addressed the impacts of dog fouling (Bull, 1998; Groome, Denton, & Smith, 2018; Taylor, Anderson, Taylor, Longden, & Fisher, 2005). Dogs will typically defecate within 10 minutes of a walk starting, and as a consequence most (but not all) deposition tends to occur within 400m of a site entrance (Taylor *et al.,* 2005). In addition, most faeces are deposited close to the path, with a peak at approximately 1m from the path edge (Shaw, Lankey, & Hollingham, 1995). Similarly, dogs will typically urinate at the start of a walk, but they will also urinate at frequent intervals during the walk too. The total volume deposited on sites may be surprisingly large. Over one year at Burnham Beeches National Nature Reserve, Barnard (2003) estimated the total amounts of urine as 30,000 litres and 60 tonnes of faeces from dogs.

3.41 Nutrient levels in soil (particularly nitrogen and phosphorous) are important factors determining plant species composition for many habitats, the typical effect will be equivalent to applying a high level of fertilizer, resulting in a reduction in species richness and the presence of species typically associated with more improved habitats. The impacts of dog fouling can often be seen in the form of grassy wedges/edges of paths on many heaths with high levels of access. This can be exacerbated by trampling, which has a lesser effect on species such as grasses (which grow from the base rather than the tip).

3.42 Ponds and small water bodies are often popular with dogs and dog walkers will often seek such features out, particularly in hot weather. Heavy use by dogs leads to turbid water, an impoverished invertebrate flora, and a loss of vegetation (Denton & Groome, 2017; Groome et al., 2018). These impacts are linked to the trampling/splashing of the dogs and are potentially exacerbated by contamination from wormer, tick, and flea treatments (Groome et al., 2018).

3.43 A further consideration is that of sunscreen and other personal care products. Personal care products containing oxybenzone and octinoxate are being banned from some areas of the world where they are thought to be contributing to the disruption of marine ecosytems. In freshwater systems, carbon-based and nano-particulate UV filters have been shown to negatively impact invertebrates (e.g. Schmitt, et al., 2008). These issues will be relevant where people enter the water.

3.44 Litter can cause direct mortality of bird, reptile, and small mammal species, and may become concentrated within certain habitats (e.g. on beaches: (Storrier, et al., 2007)). It can also be problematic for site managers if ingested by livestock.

3.45 Contamination can also include the spread of invasive species. Studies have shown that recreation can act as a vector to spread seeds over considerable distances (Wichmann et al., 2009). Additional footfall and dogs entering water bodies may increase the spread of species such as New Zealand Pigmyweed *Crassula helmsii* (Groome et al., 2018), which has been recorded near the northern boundary of the Minsmere RSPB reserve[1]. On terrestrial habitats, the occurrence of Piri-piri Burr *Acaena novae-zelandiae* has long been linked to recreation (Usher et al., 1986). The species is now widespread in the area, including around the Minsmere main RSPB car-park.

Relevant qualifying features

3.46 Relevant qualifying features are nutrient-poor habitats such as heathland and water features that are readily accessible to people and dogs. Relevant features are listed in Table 4.

[1] See RSPB website, accessed 4th July 2020

Table 4: European site qualifying features identified as being susceptible to contamination impacts from recreational activity

Qualifying feature	European site(s)	Seasonality	References	Notes
H4030 European dry heaths	Minsmere-Walberswick Heaths & Marshes SAC	Throughout year	(Denton & Groome, 2017; Marrs, 1988; Shaw et al., 1995; Underhill-Day, 2005)	Dog faeces/urine will enrich nutrient poor habitats and lead to changes in community composition. This can become concentrated car parks/access gates and can form grassy wedges alongside tracks and paths. Litter can prove a fire risk (e.g. glass, dropped cigarettes), and may prove problematic for site managers if ingested by grazing livestock. Spread of invasive species such as Piri-Piri Burr also a risk.
H1210 Annual vegetation of drift lines	Minsmere-Walberswick Heaths & Marshes SAC; Orfordness-Shingle Street SAC	Throughout year	(Storrier et al., 2007)	Risk of enrichment from dog faeces/urine. Litter (plastic in particular) often concentrated on tideline, potentially inducing physiological stress/shading if deposited over vegetation.
H1220 Perennial vegetation of stony banks	Minsmere-Walberswick Heaths & Marshes SAC; Orfordness-Shingle Street SAC	Throughout year		Risk of enrichment from dog faeces/urine.
H1150 Coastal lagoons	Benacre to Easton Bavents Lagoons SAC; Orfordness-Shingle Street SAC	Throughout year	(Saunders et al., 2000)	Risk of enrichment from dog faeces/urine. Also contamination from dog/animal skin treatments and human sunscreen. Risk of transporting disease (e.g. Chytrid) and invasive species propagules between wetland sites on footwear/clothing. Litter/waste impacts for coastal lagoons described as' observable' by Saunders *et al.*
H1330 Atlantic salt meadows	Alde-Ore & Butley Estuaries SAC	Throughout year		Risk of enrichment from dog faeces/urine, in addition to litter deposition, although severe impact considered less likely due to tidal nature of habitat.
Wetland habitats	Minsmere-Walberswick Ramsar	Throughout year		Risk of enrichment from dog faeces/urine. Also contamination from dog/animal skin treatments. Risk of transporting disease (e.g. Chytrid) and

Qualifying feature	European site(s)	Seasonality	References	Notes
				invasive species propagules between wetland sites on footwear/clothing. Litter may prove problematic for site managers if ingested by grazing livestock.
Nationally scarce plant species and British Red Data book invertebrates	Minsmere-Walberswick Ramsar	Throughout year		Risk of enrichment from dog faeces/urine. Also contamination from dog/animal skin treatments. Risk of transporting disease (e.g. Chytrid) and invasive species propagules between wetland sites on footwear/clothing. Litter may prove problematic for site managers if ingested by grazing livestock.

Impacts on site management

Overview

3.47 Damage to infrastructure can occur in a variety of ways. With more footfall, infrastructure such as car parks, paths, gates, and stiles are likely to need more maintenance and repair, and direct damage can also occur through vandalism. Increased levels of use may also lead to an increase in public opposition to management programmes associated with conservation, such as the control of invasive species (Bremner & Park, 2007). Seasonal impacts may also occur due to closures of certain paths in periods of wet weather, leading to time taken up by site managers to deal with complaints from members of the public and a potential increase in visitor numbers in novel areas of a site.

3.48 Recreational activity can be a particular problem where livestock grazing is used in conservation grazing. Dog-walkers and horse riders may be concerned about interactions with livestock, while walkers and others may be concerned about the impact of fencing and gates on open access areas or about disease. In some cases, livestock grazing (particularly sheep) is found to be untenable on sites popular with dog walkers due to worrying and death of sheep by dogs (e.g. Taylor *et al.* 2005), and displacement of livestock away from heavily utilised areas may occur. Dogs are also an issue for the welfare of livestock through the transfer of pathogens such as *Neospora* from dogs to cattle through dog faeces (causing abortion in infected cattle).

3.49 Another potential issue relates to demand for access and pressure for particular interventions, infrastructure or facilities. On sites with current recreation use visitors may well wish for better path surfacing, toilets, cafes, dog bins etc. Where access is not encouraged or there is no access there may be demand from local people and visitors for access to be provided. These issues can bring added pressure for site managers or a need to compromise between nature conservation and recreation.

Relevant qualifying features

3.50 Pressure on site managers to maintain the upkeep of recreation/access infrastructure (e.g. fencing, gates, car parks, etc) will be inherent in the face of increasing recreational activity levels, irrespective of the habitat under review. Particular focus has therefore been placed upon those habitats within which livestock are used in conservation grazing.

3.51 Ponies and cattle are used to graze areas of heathland within the SPA/SAC/Ramsar, with more robust, free-ranging, Koniks present within wetter habitats. Potential risks arise for these livestock from the ingestion of litter, worrying by dogs, and damage to fencing (leading to them accessing unsuitable areas). The presence of increased numbers, or novel, visitors within areas subject to grazing may lead to the livestock avoiding busier parts of the site (or conversely congregating if visitors begin to feed them). Any impact on the grazers could potentially impact upon the structure and quality of any of the habitats within which they are deployed.

3.52 Impacts for different European site qualifying features are summarised in Table 5.

Table 5: European site qualifying features identified as being susceptible to changes in site management arising from recreational activity

Qualifying feature	European site(s)	Seasonality	References	Notes
H4030 European dry heaths	Minsmere-Walberswick Heaths & Marshes SAC	Throughout year		Ponies, cattle and sheep used for conservation grazing on the heath and adjoining acid grassland. Damage to infrastructure and the presence of dogs/people could lead to changes in the distribution of grazers on site. Also risk of killing, injury, or disease transmission from dogs to livestock. Increased access makes some aspects of practical conservation management (e.g. tree clearance burning) more difficult.
Wetland habitats	Minsmere-Walberswick Ramsar	Throughout year		Konik ponies are used as conservation grazers in wetland areas. Damage to infrastructure and the presence of dogs/people could lead to changes in the distribution of grazers on site. Also risk of killing, injury, or disease transmission from dogs to livestock. Increased levels of access may add to pressure for water levels/flood risk to be managed differently.
Nationally scarce plant species and British Red Data book invertebrates	Minsmere-Walberswick Ramsar	Throughout year		Habitat structure will be important for these species. Konik ponies are used as conservation grazers in wetland areas. Damage to infrastructure and the presence of dogs/people could lead to changes in the distribution of grazers on site. Also risk of killing, injury, or disease transmission from dogs to livestock. Increased levels of access may add to pressure for water levels/flood risk to be managed differently.

4. Implications of Sizewell C in terms of recreation use and impact

4.1 The Suffolk coast draws visitors for a wide range of recreation use. These include a mix of tourists, day-visitors and local residents. In total, nearly 13 million visits were estimated to East Suffolk in 2018 (Destination Research, 2018), comprising a mix of day trips (just over 12 million) and staying trips (nearly 700,000).

4.2 In the Sizewell area, there are a wide range of opportunities for informal recreation use of countryside sites, these include 'flagship' locations such as the main RSPB car-park at Minsmere or the National Trust car-park at Dunwich. At these locations there are visitor centres, visitor engagement staff and facilities provided for recreation use. These locations are free for members but otherwise there is a fee to access. There are also numerous much more informal locations where parking is free and there are footpaths or open access land with relatively little access infrastructure in place.

4.3 EDF have provided some estimates of visitor numbers to different locations around Sizewell and increases in use they predict associated with the proposals for Sizewell C (i.e. additional recreational use from construction workers and from displaced visitors). The data from EDF are summarised in Maps 2 and 3[2]. It is important to note that we have found errors in the EDF reports and do not agree with the methods used (see separate review for details and comment).

4.4 The EDF estimates relate to very broad areas– i.e. 'Sandlings Walk', 'Rendlesham', and 'Minsmere Outer'. These are too ambiguous and very generalised, encompassing multiple access points and a wide area. It is impossible to drill down into the data at a scale that ecological data and visitor data can be combined in a meaningful way. Many places (e.g. Dunwich Beach, Tunstall Forest, Sandlings Walk, Rendlesham, Orford, Aldeburgh, Southwold and Walberswick) are all assumed to have the same levels of current use (150,000 per annum), which is unlikely.

[2] The maps are drawn from the data in Table 2.1, and the figures based on the 500,000 visits to Sizewell area columns (page 320 of the pdf) within Appendix E: Recreational Disturbance Assessment, within the 5.10 Shadow HRA Vol. 1: Screening and appropriate assessment. PINS document reference: APP-148

4.5 Despite these concerns, we have generated Map 2, which shows current visitor estimates by EDF and the estimates of additional use. The pie charts are graduated to reflect the total number of visitors (current and new combined) that EDF estimate. The pie charts are drawn in the approximate locations reflecting the names used by EDF and relate to a general area rather than the specific location.

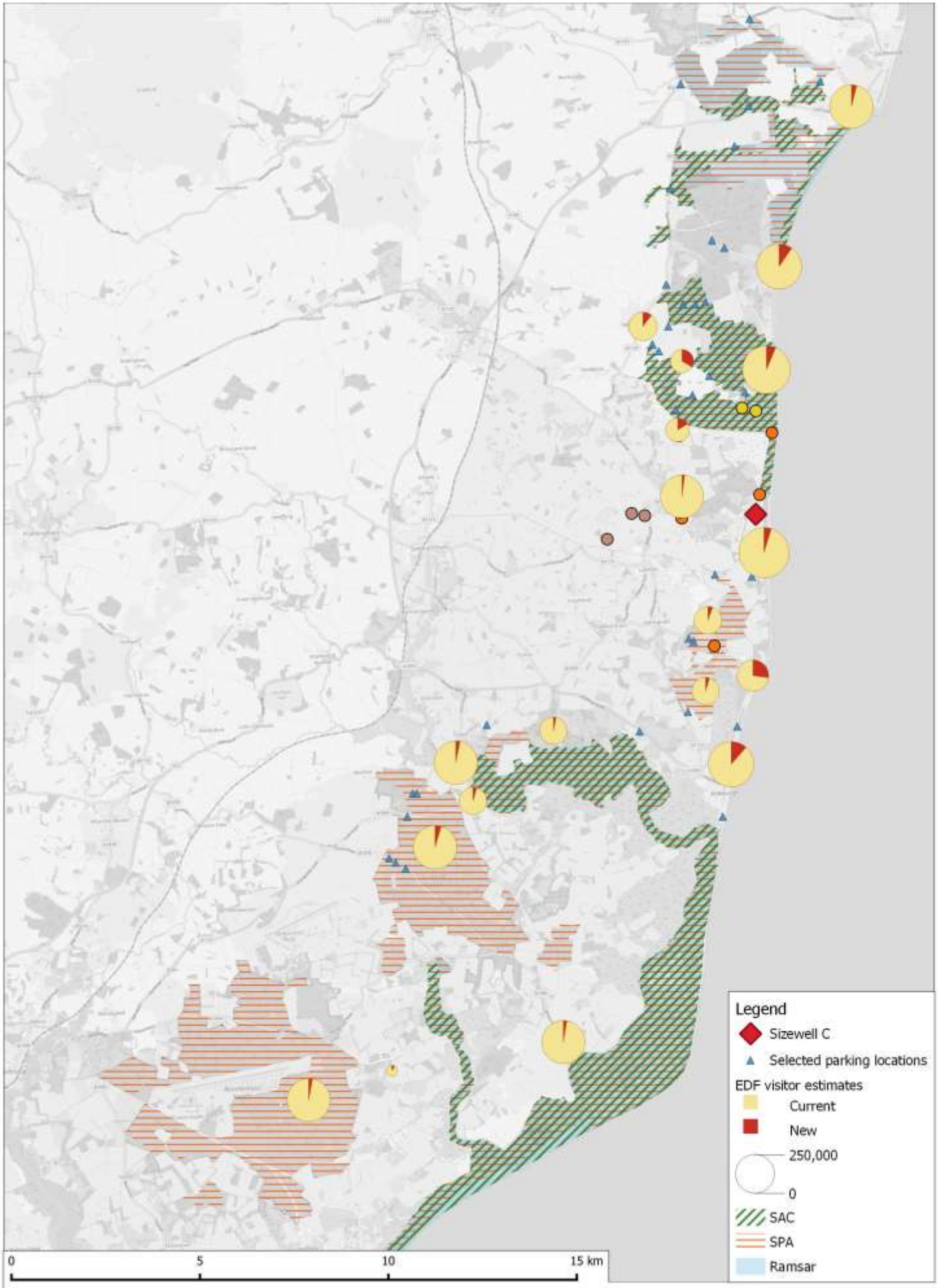
4.6 Map 3 is similar and uses the same data, although the circles are graduated to show the % increase in use estimated by EDF for each general location.

4.7 In both Map 2 and Map 3 we have also shown parking locations. These are approximate and intended as a guide only, they have not been systematically mapped on the ground and checked. These blue points do highlight the range of parking locations whereby visitors can access European sites. We would have expected EDF to present visitor data at this kind of resolution, i.e. current and predicted visitor numbers at each access point. This would have allowed much more detailed analysis of visitor data alongside ecological data (habitats and species distributions). Such data gathering would then have allowed changes in access to be related to vulnerable species, providing a means by which impacts could be properly assessed and appropriate mitigation established. Unfortunately, such data are lacking, and it is not possible to gain a robust and reliable overview of current recreation use, let alone reliable predictions of future use were Sizewell C to go ahead.

European sites and qualifying features

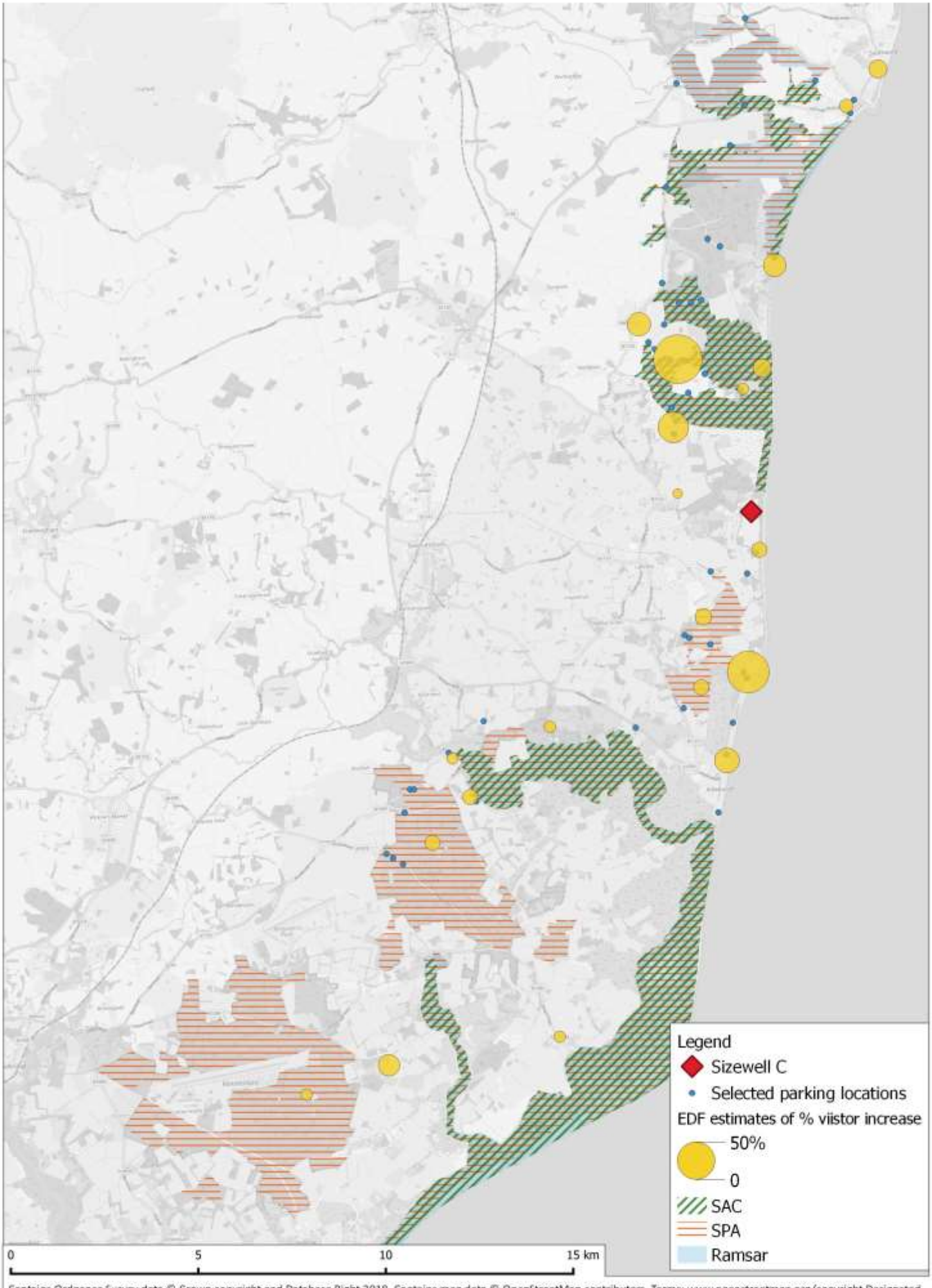
4.8 In Table 6 we summarise the European sites and identify the qualifying features we consider potentially vulnerable to recreation impacts. We have scored each according to the likely risks from the Sizewell C proposal.

Map 2: EDF estimates of visitor use and change



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Map 3: EDF estimates of visitor use: percentage increase



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Table 6: Overview of potential impacts by site and qualifying feature. All features considered at risk are listed below, where there is overlap between Natura 2000 site features and Ramsar then we have not repeated the relevant information for the Ramsar sites. We have scored the potential impacts on a scale of 1-5. 1= risks very low, likely impact low and of relatively low concern – likely significant effects (LSE) can probably be ruled out but further checks/review necessary. 5 = risks marked, likely to occur and of major concern, may be hard to rule out adverse effects on integrity, even with mitigation. Blank cells indicate no risk identified. Shading reflects the score (from 1 (amber) to 5 (dark red)). Avian interest features are identified as breeding (br) and/or wintering (w).

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
Alde, Ore & Butley Estuaries SAC	1130 Estuaries		1				These are intertidal and subtidal habitats. Access to estuary habitat at Slaughden, Snape and Iken, potentially further afield (e.g. Orford, Boyton). Risks relate to very limited trampling damage, e.g. access to boats, paddleboards, kayaks etc. Probably can be ruled out as no LSE.
	1140 Mudflats and sandflats not covered by water at low tide		1				These are intertidal habitats. Access possible at Slaughden, Snape and Iken, potentially further afield (e.g. Orford, Boyton). Risks relate to very limited trampling damage, e.g. access to boats, paddleboards, kayaks etc. and people walking out from shore. Probably can be ruled out as no LSE due to nature of terrain relatively robust habitat. Probably can be ruled out as no LSE.
	1330 Atlantic salt meadows (Glauco-Puccinellietalia)		1				Trampling damage possible on saltmarshes, which are relatively easy to walk on. Trampling can cause loss of vegetation and lead to risks from erosion (e.g. storm surges). Risks limited to locations such as Slaughden, Snape and Iken, potentially further afield (e.g. Orford, Boyton).
Benacre to Easton Bavents Lagoons SAC	1150 Coastal lagoons		1		1		Some distance away from Sizewell and therefore unlikely to see marked changes in access. Lagoons vulnerable to damage from trampling around edge and increased turbidity and pollution from dogs and people entering the water. Benacre, Covehithe, and Easton Broad are the key locations. Access likely to have little impact given dynamic coastline and changes to these habitats in recent years. Probably can be ruled out as no LSE.
Orfordness - Shingle Street SAC	1150 Coastal lagoons		1		1		These are percolation lagoons and have changed shape markedly over time. Key location is at Shingle Street, which is a considerable travel distance from the Sizewell area. Risks relate to damage from

Impacts of recreation related to Sizewell C and implications for European sites.

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
							trampling around edge and increased turbidity, and pollution from dogs and people entering the water. Starlet Sea Anemone *Nematostella vectensis* likely to be vulnerable to trampling. LSE can probably be ruled out due to the travel distance.
	1210 Annual vegetation of drift lines		1	1	1		Accessible areas at Slaughden and Shingle Street. Risks relate to trampling leading to loss of vegetation. Localised impacts from barbeques and fires on the beach, resulting in small burnt patches. Contamination from dog fouling, and possibly from litter, fishing wire etc. as a result of increased fishing use.
	1220 Perennial vegetation of stony banks		2	1	1		Accessible areas at Slaughden and Shingle Street. Risks relate to trampling leading to loss of vegetation and damage to lichens. Contamination from dog fouling, and possibly from litter, fishing wire etc. as a result of increased fishing use.
Minsmere to Walberswick Heaths & Marshes SAC	1210 Annual vegetation of drift lines		3	1	3		Concerns would relate to a marked and prolonged spike in access, at Sizewell, from Eastbridge to the Sluice area, around Dunwich Cliffs, Dunwich and Walberswick. Potential for increased use in evenings and early mornings due to proximity to construction campus. Further trampling will lead to vegetation loss, potential for beach fires/barbeques to also cause localised vegetation damage. Contamination from dog fouling.
	4030 European dry heaths		2	4	3	2	Numerous easily accessible locations where access likely to increase markedly. Trampling damage from increased footfall likely to lead to track widening, increase in grasses, exacerbated further by nutrient enrichment from dog fouling. Risks of spread of species such as Piri-Piri Burr. Fire risk marked concern, from discarded cigarettes, barbeques and campfires and potential for major incident in area between Westleton and Eastbridge or Dunwich Heath. Given proximity of construction workers campus, barbeques etc in evenings a clear risk. Impacts to management relate to the potential for grazing to be compromised, for example through incidents with dogs.

Impacts of recreation related to Sizewell C and implications for European sites.

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
	1220 Perennial vegetation of stony banks		3	2	3		Concerns would relate to a marked and prolonged spike in access, at Sizewell, from Eastbridge to the Sluice area, around Dunwich Cliffs, Dunwich and Walberswick. Potential for increased use in evenings and early mornings due to proximity to construction campus. Further trampling will lead to vegetation loss, potential for beach fires/barbeques to also cause localised vegetation damage. Contamination from dog fouling, and possibly from litter, fishing wire etc. as a result of increased fishing use.
Alde-Ore Estuary SPA	Marsh Harrier (br)	1					Disturbance at nest a risk and birds can nest in small patches of reed and crops, potentially accessible and vulnerable, e.g. from dogs. Due to distance and habitat risks probably low.
	Lesser Black-backed Gull (br)	1					Gulls breed on Orfordness and accessibility limited and access difficult. Risks very low and LSE can probably be ruled out.
	Ruff (w)	1					Low numbers winter and use wet grassland that is difficult to access due to ditches and water table. Very low risk from dogs off lead. Relevant locations will be Hollesley Marshes, Boyton Marshes, and Town Marshes, all of which are outside the SPA (but functionally-linked).
	Avocet (br & w)	2					Breeding can occur, or is possible, at a range of locations, including Hazlewood Marshes, saltmarsh areas, Havergate Island and lagoons near Shingle Street. Most areas inaccessible and access controlled, but impacts possible at some of the breeding sites. Estuary is narrow and issues for birds feeding on intertidal areas from dogs off-lead, people straying onto saltmarsh.
	Little Tern (br)	2					Little Terns have bred at a range of locations along the coast – numbers can fluctuate and colonies shift between years. Birds will shift between different European sites and different parts of the coast. Species highly vulnerable to disturbance from dogs off lead and people on beaches. Alde-Ore sites are primarily at Orfordness, where access is difficult.
	Sandwich Tern (br)	1					Main breeding site is Havergate where nesting in recent years only sporadic. Colonies can shift markedly over time and are very vulnerable to disturbance events. LSE to the species at Havergate can probably be ruled out.

Impacts of recreation related to Sizewell C and implications for European sites.

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
	Redshank (w)	2					Increased access at Iken, Snape, Hazlewood Marshes, Town Marshes and Slaughden has potential to increase amount birds are flushed and disrupt foraging. Dogs off-leads likely to be a particular issue. Impacts during particularly cold weather perhaps most likely.
Benacre to Easton Bavents SPA	Bittern (br)	1					Nest sites unlikely to be vulnerable due to difficulties in access. Birds feeding around peripheral areas of reedbeds perhaps vulnerable to increased access, including dogs off-leads. Given distance from Sizewell, LSE can probably be ruled out.
	Marsh Harrier (br)	1					Disturbance at nest a risk and birds can nest in small patches of reed and crops, potentially accessible and vulnerable, e.g. from dogs. Due to distance and habitat risks probably low.
	Little Tern (br)	2					Little Terns have bred at a range of locations along the coast – numbers can fluctuate and colonies shift between years. Birds will shift between different European sites and different parts of the coast. Species highly vulnerable to disturbance from dogs off lead and people on beaches. Main breeding at Benacre but could nest at other locations, however in recent years main colony in area is further north still at Kessingland. While highly vulnerable to disturbance impacts, concern low due to distance from Sizewell.
Minsmere to Walberswick SPA	Shoveler (br & w)	2				1	Present on pools and waterbodies and grazing marsh across the area, and potentially vulnerable to disturbance from dogs and people in the vicinity of Eastbridge, including Minsmere Levels. Impacts from site management relate to issues with achieving necessary grazing.
	Teal (br)	2				1	Present on pools and waterbodies and grazing marsh across the area, and potentially vulnerable to disturbance from dogs and people in the vicinity of Eastbridge, including Minsmere Levels. Impacts from site management relate to issues with achieving necessary grazing.
	Gadwall (br & w)	2				1	Present on pools and waterbodies and grazing marsh across the area, and potentially vulnerable to disturbance from dogs and people in the vicinity of Eastbridge, including Minsmere Levels. Impacts from site management relate to issues with achieving necessary grazing.

Impacts of recreation related to Sizewell C and implications for European sites.

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
	European White-fronted Goose (w)	3				1	Birds wintering on Minsmere Levels and nearby grazing marsh at risk from increased use by people and dogs. Risk of birds using the area less and increased flushing when present. Impacts from site management relate to issues with achieving necessary grazing.
	Bittern (br)	2		1			Nest sites unlikely to be vulnerable due to difficulties in access. Birds feeding around peripheral areas of reedbeds perhaps vulnerable to increased access, including dogs off-leads. Potential for birds to use areas such as around Eastbridge less. Fire risk to upper parts of reedbeds and during periods of extreme dry weather. Reedbed adjoins heath/dry areas e.g. Keeble Marsh and risk of fire spreading between habitats.
	Nightjar (br)	4		3		1	Birds around Westleton, on Dunwich Heath and in Dunwich Forest vulnerable to increased disturbance, particularly from dog walking. Potential for birds to avoid nesting in some areas and risk of reduced breeding success. A major fire incident on the heaths could result in direct loss of nests and reduction in number of territories in subsequent year(s).
	Marsh Harrier (br)	2		1			Disturbance at nest a risk and birds can nest in small patches of reed and crops, potentially accessible and vulnerable, e.g. from dogs.
	Hen Harrier (w)	2		2			Increased disturbance at roost could result in roost being abandoned, access in evening (e.g. dog walkers) particularly of concern. Roosts in deep heather on the heath, which could be destroyed by fire.
	Avocet (br)	3					Birds nesting on the lagoons at Walberswick marshes and Dingle vulnerable to disturbance, particularly from dogs off leads. Risks include birds avoiding areas of otherwise suitable habitat and chicks being killed by dogs.
	Little Tern (br)	3					Little Terns have previously bred at a range of locations along the coast including in front of Sizewell A Power Station. Numbers can fluctuate and colonies shift between years. Birds will shift between different European sites and different parts of the coast. This species highly vulnerable to disturbance from dogs off lead and people on beaches. Increased access could result in birds avoiding otherwise suitable habitat, colonies deserting, breeding success being compromised and dogs killing chicks.

Impacts of recreation related to Sizewell C and implications for European sites.

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
Sandlings SPA	Nightjar (br)	4		3		1	Birds around Aldringham Walks, North Warren, and potentially Snape and Tunstall Forest vulnerable to increased disturbance, particularly from dog walking. Potential for birds to avoid nesting in some areas and risk of reduced breeding success. A major fire incident could result in direct loss of nests and reduction in number of territories in subsequent year(s).
	Woodlark (br)	4				2	Birds around Aldringham Walks, North Warren, and potentially Snape and Tunstall Forest vulnerable to increased disturbance, particularly from dog walking. Also birds around Minsmere and Dunwich Forest functionally-linked to this SPA. Potential for birds to avoid nesting in some areas and risk of reduced breeding success. A major fire incident could result in direct loss of nests and reduction in number of territories in subsequent year(s). Access may also compromise ability to graze sites effectively e.g. through dog incidents and livestock. Marked increases in dogs may affect rabbit distribution and use, which is also important for Woodlarks.
Alde - Ore Estuary Ramsar (additional to SPA/SACs)	Notable assemblage of breeding and wintering wetland birds	2					Increased access at Iken, Snape, Hazlewood Marshes, Town Marshes and Slaughden has potential to increase amount birds are flushed and disrupt foraging. Dogs off-leads likely to be a particular issue.
Minsmere - Walberswick Ramsar (additional to SPA/SAC)	Site contains a mosaic of marine, freshwater, marshland and associated habitats, complete with transitional areas in between. Contains the largest continuous stand of reedbeds in England		1	2	2	2	Trampling damage in drier areas and peripheral habitats possible. Major fire incident in reedbeds a concern. Contamination risks from dogs in ditches, pools, and wet habitats spreading invasive species such as New Zealand Pygmyweed. Dog incidents could affect ability to graze effectively.

European site(s)	Qualifying feature	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes and key concerns
	and Wales, and rare transition in grazing marsh ditch plants from brackish to fresh water						
	Nationally scarce plant species and British Red Data book invertebrates, including a population of Narrow-mouthed Whorl Snail		1	1	2	1	Trampling damage in drier areas and peripheral habitats possible. Major fire incident in reedbeds a concern. Contamination risks from dogs in ditches, pools and wet habitats spreading invasive species such as New Zealand Pygmyweed. Dog incidents could affect ability to graze effectively.
	An important assemblage of rare breeding birds associated with marshland and reedbeds	2		1		1	Risks from increased disturbance around periphery of wetland areas, from dogs off-leads and walkers. A major fire incident in reedbed could result in loss of breeding territories and species richness. Dog incidents could affect ability to graze effectively.

Vulnerable features that are not qualifying interest

4.10 Our focus in this report is on the European site interest. However, it is important to note that the area contains a wide range of habitats and species that is of outstanding importance for nature conservation and unique within the UK. There are other species and habitats in the vicinity which, while not qualifying features for the European site, are also of considerable conservation importance and potentially vulnerable. These include:

- **Stone Curlew** *Burhinus oedicnemus* breed on the acid grassland and open sandy farmland in the Walberswick/Westleton area, including the outer parts of the Minsmere RSPB reserve. The population is relatively low and the species is particularly vulnerable to disturbance, with dogs off-led a concern (Clarke & Liley, 2013; Taylor, 2006; Taylor, Green, & Perrins, 2007).
- **Ringed Plover** nest in small numbers on the beaches and are highly vulnerable to disturbance, which influences settlement pattern and breeding success (Liley & Sutherland, 2007).
- **Dartford Warbler** *Sylvia undata* nest low down in heathland vegetation and studies have shown impacts from disturbance, with reduced breeding success in disturbed territories. This species returned to Suffolk in the 1990s.
- **Adder** *Vipera berus* are found in a range of habitats in the general area, including heathland, scrub and the back of the beach. This declining reptile is vulnerable to recreation impacts, particularly dog attacks (Worthington-Hill, 2015)**.**
- **Acid Grassland** is vulnerable to impacts from fire and eutrophication (e.g. from dog fouling), while extensive grazing which is important to maintain the habitat, can be difficult where access is high.
- **Scarce invertebrates** there are a number of species, such as Antlion *Euroleon nostras*, for which the Minsmere area is important. There are risks for some of these species from access, for example through trampling and fire.

5. Mitigation

5.1 In this section we briefly consider the likely visitor profile and behaviours and set out broad approaches to mitigation that might be relevant.

Visitor profile and behaviours

5.2 The issues identified arise from increased recreation use associated with a marked increase in the local population and displacement of existing access.

5.3 It is important to recognise that there is potential for a marked change in visitor profile and the way people access the sites identified. This is because use by locally based construction workers will be very different to the use associated with recreation by tourists, day visitors travelling from a wide area, and second homeowners. Postcode analysis of visitors to the main part of the Minsmere RSPB Reserve (e.g. Hoskin et al., 2019) reveals people travelling considerable distances to visit the reserve. These visitors are likely to be drawn for the specific nature interest and wildlife spectacle.

5.4 People coming to the area to live and work in the vicinity will seek nearby greenspace for recreational use that is not necessarily related or connected to the area's conservation interest, and such use will include exercise and dog walking. It may extend to sea angling, barbeques, swimming, and beach recreation. Such people are likely to consciously avoid the areas they perceive as 'nature reserves' (i.e. where there are hides, wardens and a fee to enter) and instead focus their use at more peripheral locations. These are often still part of the European site and do not have necessary infrastructure to manage access. For example, around the Westleton and Eastbridge there are numerous public rights of way and also many areas that are 'open access' under the Countryside and Rights of Way Act (2000). The focus for increased recreational use is likely to be away from the main car-parks (such as the main Minsmere car-park and NT Dunwich Heath) where existing visitor management and engagement is focussed. We would expect construction workers based in the area to utilise more informal parking and locations where they can easily access beaches, quiet areas of heath, etc.

5.5 Use by construction workers will also be likely to take place in the early mornings, evenings, and around shift work and therefore not necessarily fit with peak visitor use and the current visitor profile.

Mitigation Options

5.6 Given the range of issues identified and the range of risks, a package of mitigation measures will be necessary. Any package will need to involve a number of different measures and be flexible in order to respond to changing access patterns or particular issues. Weather conditions, trends in access, the occurrence of rare wildlife, and a range of other factors will influence the scale of risk and the particular concerns at any one time. Different measures may be more important at some times compared to others. For example, prolonged periods of dry weather and a hot summer will bring particular fire risks, while cold winter weather may make birds more vulnerable to disturbance.

5.7 An overview of some potential mitigation options are summarised in Table 7. This indicates which measures may relate to which kinds of impact. In order to develop any mitigation package in detail, it will be necessary to draw on visitor data and liaise closely with the relevant site managers.

5.8 Any mitigation package should also include comprehensive monitoring that should be integrated alongside the mitigation and pick-up any emerging issues. Monitoring would potentially include counts of parked vehicles, counts of people, visitor interviews and regular checks on infrastructure. Unfortunately, the data collection to inform the evidence base presented by EDF is inadequate to form a foundation for such monitoring. Ideally the data collected to date would provide a baseline from which changes could be identified and locations targeted for mitigation. Unfortunately EDF have not collected such data at the relevant level of detail.

Table 7: Overview of possible mitigation options.

Mitigation option	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes
Increased warden presence	✓	✓	✓	✓	✓	Targeted to vulnerable locations. Can be flexible and responsive to emerging issues. Needs to be adequately resourced to be effective as mitigation and likely to involve anti-social hours.
Fenced exclosures on beach	✓	✓	✓			Not always effective and need to be combined with other measures (signage, warden presence). Fencing can protect vegetation and suitable habitat for breeding birds.
Access restriction	✓	✓		✓	✓	Options include enhancing fencing, ensuring gates are locked, making gates effective barriers for dogs (e.g. with mesh), blocking access to watersides etc. These measures provide potential to restrict access by dogs into sensitive watercourses, access onto grazing marsh etc. These approaches can be difficult to maintain and can be compromised by vandalism, cutting etc.
Temporary signage	✓	✓	✓	✓		Providing ability to highlight particular issues, such as high fire risk, presence of rare breeding birds, livestock.
Waymarking and permanent signage	✓	✓		✓	✓	Helps direct visitors and manipulate visitor flow.
Path management	✓	✓			✓	Boardwalks and vegetation management on path edges can help keep access to paths. Potential for less sensitive public footpaths and routes to be enhanced.

Mitigation option	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes
Dead hedging	✓	✓			✓	For example, using cut gorse and brash. Potential to close off desire lines as they form and potentially limit access by dogs. Likely to be effective only as a short term response measure and reactive.
Firebreaks			✓			Additional firebreaks, potentially targeting protection of areas at risk for changes in access. Can create new desire lines so need to be chosen carefully. Regular cutting required to maintain.
Fire management planning			✓			Reserves will have existing fire management plans. These may benefit/need reviewing in light of changes in access and visitor pressure.
Access to water/provision of new fire hydrants			✓			New fire hydrants on heath have been provided as mitigation for housing on Dorset Heaths. Clearly will not stop impact occurring but may lessen effect.
Public engagement	✓		✓	✓		Scope for messaging to target particular behaviours and issues, such as as use of disposable barbeques, dogs off-leads, dog fouling etc. Potential to react to new/emerging issues. Potential for targeted information provision at construction campus as to where to go for recreation. There is little certainty associated with these measures and they perhaps work best to reinforce other measures (e.g. wardening).
Controls on local shops selling disposable barbeques			✓			Potentially difficult to secure and limited in effectiveness if available more widely.

Mitigation option	Disturbance	Damage	Fire	Contamination	Impacts on site management	Notes
Controls on parking	✓	✓	✓	✓	✓	Potential to close off small parking locations, e.g. with bunds or dragons teeth. Also possible to reduce the number of spaces. Gates on car-parks can prevent access in late evening/early morning and help limit anti-social behaviour.
Provision of dog bins/litter bins				✓		Bins require emptying and therefore cost to maintain.
Provision of dedicated space for exercise and dog walking for construction workers	✓	✓		✓	✓	Would need to be large and could be promoted directly to workers. May be difficult to provide an adequate space to deflect people from the coast.
Gym membership for construction workers	✓	✓				Would not be effective for dog walkers but may work to direct certain activities (e.g. jogging) away from sensitive locations. Would require some research and survey work to determine whether any likelihood of effectiveness.
Screening	✓					Potential for planting/vegetation growth, use of reed/heather screens or other structures to create visual barriers for people and dogs. Likely to be an option only in limited/specific locations.
Manipulation of water levels	✓	✓				Raising water levels or retaining deep water in ditches etc. may help restrict access to some locations.
Enforcement	✓			✓		There may be scope with PSPOs in some locations to control dogs off leads and fouling. Would require enforcement and may be difficult to establish.

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FOOTPRINT ECOLOGY



Review of Sizewell C application documents and evidence in relation to recreation impacts

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Summary

Footprint Ecology have been commissioned by the National Trust and RSPB to provide a review of evidence (submitted as part of the Sizewell C application) on recreation issues. The construction of the power station will require a peak workforce of 7,900 construction workers, leading to a marked increase in the number of residents in the area. In addition, construction will result in the closure of access routes and the displacement of visitors due to path closures, noise, additional traffic, etc. Marked changes in local access are therefore likely. The Sizewell C site is directly adjacent to multiple European sites where there are existing rights of access and numerous species and habitats vulnerable to recreation impacts.

Recreation impacts include disturbance, trampling, increased fire risk, and contamination (e.g. spread of invasive species, dog fouling, etc). Some of the key sensitive areas are already carefully managed, with resources in place to both manage access and allow visitors to experience the area while minimising harm. Areas such as the main car park at Minsmere draw visitors from across the country and are well-known visitor destinations, drawing visitors for the wildlife spectacle and nature interest. Here there are well marked paths, hides, screens, wardens and careful management to allow wildlife to flourish and people to be able to experience it.

Concern with respect to the Sizewell C proposals and recreation impacts therefore relates to vulnerable features within the more peripheral areas of the European sites, where public footpaths, informal parking, and open access provide opportunities for recreational use, and where any issues arising from access are much harder to manage. Access to these locations is more likely to be for dog walking, jogging, picnics, barbeques, fishing, etc. (rather than to see wildlife).

Our review highlights that:

The baseline for current visitor use is inadequate. Count data (visitor numbers) are only provided for a small number of locations (surveyed in 2014) and just for August and November. Many of the smaller, more vulnerable, locations (from an ecological perspective) have not been surveyed.

Predictions of increased use are confusingly presented, with different estimates given in different reports alongside some clear errors. Estimates appear very low compared to national data and are much lower than we would expect, based on our experience of visitor surveys and recreational use of the countryside.

Impacts are poorly assessed, and the consideration of impacts for the following are particularly poor and inadequate:

- Perennial vegetation of stony banks (Minsmere to Walberswick Heaths & Marshes SAC);
- European Dry Heaths (Minsmere to Walberswick Heaths & Marshes SAC);
- Little Tern (Minsmere-Walberswick SPA);
- Nightjar (Minsmere-Walberswick SPA);
- Hen Harrier (Minsmere-Walberswick SPA);
- Wintering waterbirds (Shoveler, Gadwall, White-fronted Goose) (Minsmere-Walberswick SPA);
- Breeding waterbirds (Shoveler, Gadwall, Teal) (Minsmere-Walberswick SPA), and;
- Woodlark (Sandlings SPA).

Mitigation measures are scant and fail to address the nature conservation issues that are likely to arise.

Ultimately the Suffolk Coast is of unique importance for nature conservation and supports a diversity of habitats and species that is unparalleled in the UK, particularly in the area between Slaughden and Southwold. The area also draws high numbers of visitors, resulting in a difficult balancing act to manage both access and nature conservation; it is a crowded area.

Sizewell C would disrupt the balance and the application documents do not provide the robust assessment and mitigation package that we expected. The assessment of impacts is inadequate, and predictions of changes in visitor use are full of inaccuracies, confusing, and poorly put together. Necessary mitigation, targeted to the vulnerable locations, is virtually non-existent. Ultimately, the information as presented in the Shadow HRA does not allow a conclusion of no adverse effect on integrity, from recreation impacts, to be drawn for a range of different species and habitats across multiple European sites.

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1. Introduction

1.1 This document has been commissioned by the National Trust and RSPB to assist both organisations with their response to the application for a new power station at Sizewell C. Both organisations are concerned about impacts from recreation, for example through displacement of recreational users from the vicinity of the Sizewell C development to neighbouring sites during the construction period.

1.2 This report involves a review of specific ecological reports with the aim of checking the robustness of EDF's conclusions. We structure the report addressing the following questions, set by the National Trust/RSPB:

- What is Footprint Ecology's view on the adequacy of EDF's assessment of recreational disturbance impacts (particularly in the HRA) and the robustness of the data on which this is based? Please qualify this view.
- Is any monitoring strategy proposed by EDF sufficient to detect ecological impacts of increased recreational disturbance, with appropriate triggers to secure further mitigation if adverse effects are shown, and assess the effectiveness of mitigation? This may be included in the DCO application or be developed as an Examination requirement.
- Is any mitigation strategy proposed by EDF sufficient to address ecological impacts of increased recreational disturbance? Again, this may be included in the DCO application or be developed as an Examination requirement.
- What would be valuable questions and points that National Trust and RSPB should raise in relation to recreational disturbance impacts with EDF and why?

1.3 More general points or concerns are set out in a table at the end of the report, which picks up any additional points that result from our review.

1.4 Our focus is on the recreation impacts to the nature conservation interest, particularly that relating to European sites. Table 1 lists the documents referred to in this report.

Table 1: Documents referred to in this report. Hyperlinks are provided in the final column.

Name of document (as used in this report)	Full document title	Filename	Page number of pdf where doc starts (as relevant)	Doc number
Shadow HRA	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	EN010012-001765-SZC_Bk5_5.10_V1_Shadow_HRA_Report_Part_1_of_5		APP-145
Recreational Disturbance Assessment	Shadow HRA report Appendix E: Recreational Disturbance Assessment	EN010012-001768-SZC_Bk5_5.10_V1_Shadow_HRA_Report_Part_4_of_5	305	APP-148
Shadow HRA Recreational Disturbance Evidence Base	SHADOW HRA ANNEX A: RECREATIONAL DISTURBANCE EVIDENCE BASE	EN010012-001768-SZC_Bk5_5.10_V1_Shadow_HRA_Report_Part_4_of_5	397	APP-148
Terrestrial Ecology and Ornithology Report	6.3 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology	EN010012-001844-SZC_Bk6_ES_V2_Ch14_Terrestrial Ecology and Ornithology		APP-224
Sport and Leisure Audit	Volume 2, Main Development Site, Chapter 9, Appendix 9E: Sport and Leisure Audit and Estimated Demand	SZC_Bk6_ES_V2_Ch9_Socio-economics_App9A_9f	75	APP-196
Terrestrial Ecology and Ornithology Report – Ornithology Appendix	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14A7: Ornithology Part 1 of 2	EN010012-002256-SZC_Bk6_ES_V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14A7_Orni_2013117_1		APP-237
Plants and Habitats Synthesis Report	6.3 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14B1 Plants and Habitats Synthesis Report	EN010012-001871-SZC_Bk6_ES_V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14B1_Plants_Habitats_Synthesis		APP-250

Name of document (as used in this report)	Full document title	Filename	Page number of pdf where doc starts (as relevant)	Doc number
Ornithology Synthesis Report	6.3 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14B2 Ornithology Synthesis Report	EN010012-001856-SZC_Bk6_ES_V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14B2_Ornithology_Synthesis		APP-251
Amenity and Recreation Report	6.3 Volume 2 Main Development Site Chapter 15 Amenity and Recreation	EN010012-001882-SZC_Bk6_ES_V2_Ch15_Amenity and Recreation		APP-267
2014 Sizewell C Visitor Surveys	6.3 Volume 2 Main Development Site Chapter 15 Amenity and Recreation Appendix A: 2014 Sizewell c visitor surveys	EN010012-001884-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_1_of_3		APP-268
2015 Sizewell C RSPB Minsmere Visitor Surveys	6.3 Volume 2 Main Development Site Chapter 15 Amenity and Recreation Appendix B: 2015 Sizewell c RSPB Minsmere visitor surveys	EN010012-001885-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_2_of_3	2	APP-269
2016-2018 Sizewell C Visitor Surveys	6.3 Volume 2 Main Development Site, Chapter 15 Amenity and Recreation Appendix15C: 2016-2018 SIZEWELL C VISITOR SURVEYS 2	EN010012-001885-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_2_of_3	63	APP-269
Amenity and Recreation Baseline Report	ES VOLUME 2, CHAPTER 15, APPENDIX 15F: AMENITY AND RECREATION BASELINE REPORT	EN010012-001886-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_3_of_3	43	APP-270
Description of non-significant effects	6.3 Volume 2 Main Development Site Chapter 15 Amenity and Recreation Appendix 15G: Description of non-significant effects	EN010012-001886-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_3_of_3	78	APP-270
Rights of Way and Access Strategy	VOLUME 2, CHAPTER 15, APPENDIX 15I: RIGHTS OF WAY AND ACCESS STRATEGY	EN010012-001886-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_3_of_3	111	APP-270

Name of document (as used in this report)	Full document title	Filename	Page number of pdf where doc starts (as relevant)	Doc number
Planning Statement	Planning Statement Appendix 8.4J Section 106 Heads of Terms	EN010012-002218-SZC_Bk8_8.4_Planning_Statement_AppxJ_S106_Heads_of_Terms		APP-600
Mitigation Route Map	8.12 Mitigation Route Map	EN010012-002234-SZC_Bk8_8.12_Mitigation_Route_Map		APP-616

About Footprint Ecology

1.5 Footprint Ecology are a specialist consultancy with particular expertise in the assessment of recreation impacts on nature conservation sites. Our work includes Local Plan HRAs, writing mitigation plans for European sites, visitor survey work, reserve management plans, and visitor management strategies for nature conservation sites.

1.6 Our work on visitor management, recreation assessment, and visitor surveys has included sites such as the New Forest National Park, Epping Forest, Burnham Beeches, the North Kent coast, Wicken Fen, Hatfield Forest, the Humber, Morecambe Bay, the Severn Estuary, Poole Harbour, the Exe Estuary, the Solent, the Dorset Heaths, the Wealden Heaths and the Thames Basin Heaths.

1.7 We have provided advice to Natural England on open access provision under CRoW and on coastal access and we have acted for Natural England, RSPB and local authority clients at public inquiry. Previous work in Suffolk has included visitor surveys on the southern part of the Suffolk Sandlings, a visitor survey of the River Deben, Local Plan HRA work for Ipswich Borough Council, Suffolk Coastal District Council, and Waveney District Council (and now East Suffolk Council). We wrote the Suffolk Coast Recreational Disturbance Avoidance and Mitigation Strategy (RAMS).

2. Adequacy of EDF's assessment of recreation impacts

Overview

2.1 The assessment by EDF is not adequate to rule out the conclusions of no adverse effect on integrity set out in the HRA.

2.2 We expected to see detailed visitor survey work that provided an accurate description of the current use, showing numbers of visitors and footfall across the relevant European sites and supporting habitat. This would have provided the detail of levels of use along different paths and the number of vehicles parked in different locations, allowing maps of current recreation use to be produced. Change could then be predicted to show how construction work and visitor displacement would change access and maps produced to show change alongside ecological data (species and habitats). This would then clearly indicate which locations and which interest features would be vulnerable to change and ensure mitigation measures were fit for purpose and appropriately targeted.

2.3 Instead we find a number of errors and muddled information, with conflicting information scattered across different reports. There is a lack of detail as to what levels of change are to be expected in which precise locations and a failure to apply this to the ecological interest at the appropriate level of detail.

Baseline on visitor use

Existing visitor data

2.4 The **Amenity and Recreation Baseline Report** (para 1.2.2) and **Shadow HRA recreational disturbance evidence base** (para 2.4.1) both list a range of studies as relevant baseline visitor survey information.

2.5 It is noteworthy that while the list in the Shadow HRA recreational disturbance evidence base and the Amenity and Recreation Baseline in the ES are superficially similar, they differ, with 13 studies listed in the Shadow HRA recreational disturbance evidence base and 11 in the Amenity and Recreation Baseline Report, and a different mix in each. While the Shadow HRA recreational disturbance evidence base supposedly includes reports to 2019 and the Amenity and Recreation Baseline only to 2018, it is only the list

in the Amenity and Recreation Baseline that includes a report from 2019. These errors highlight that material is repeated in different places and shoddily collated and does not provide confidence of a robust assessment.

2.6 The **Amenity and Recreation Baseline Report** (para 1.2.4) describes this as a 'good resource'. However, most of the studies listed are very specific (e.g. RSPB data for the core part of Minsmere Reserve) or are general (e.g. Suffolk County Council's Green Access Strategy) with limited relevance. Most are dated (e.g. 2010, 2011, even 2004) and most cover locations well away from Sizewell (e.g. Deben Estuary). As a baseline there is scant information on visitor numbers, activities at different locations relevant to Sizewell C, or how use varies at the different sites.

Surveys commissioned by EDF

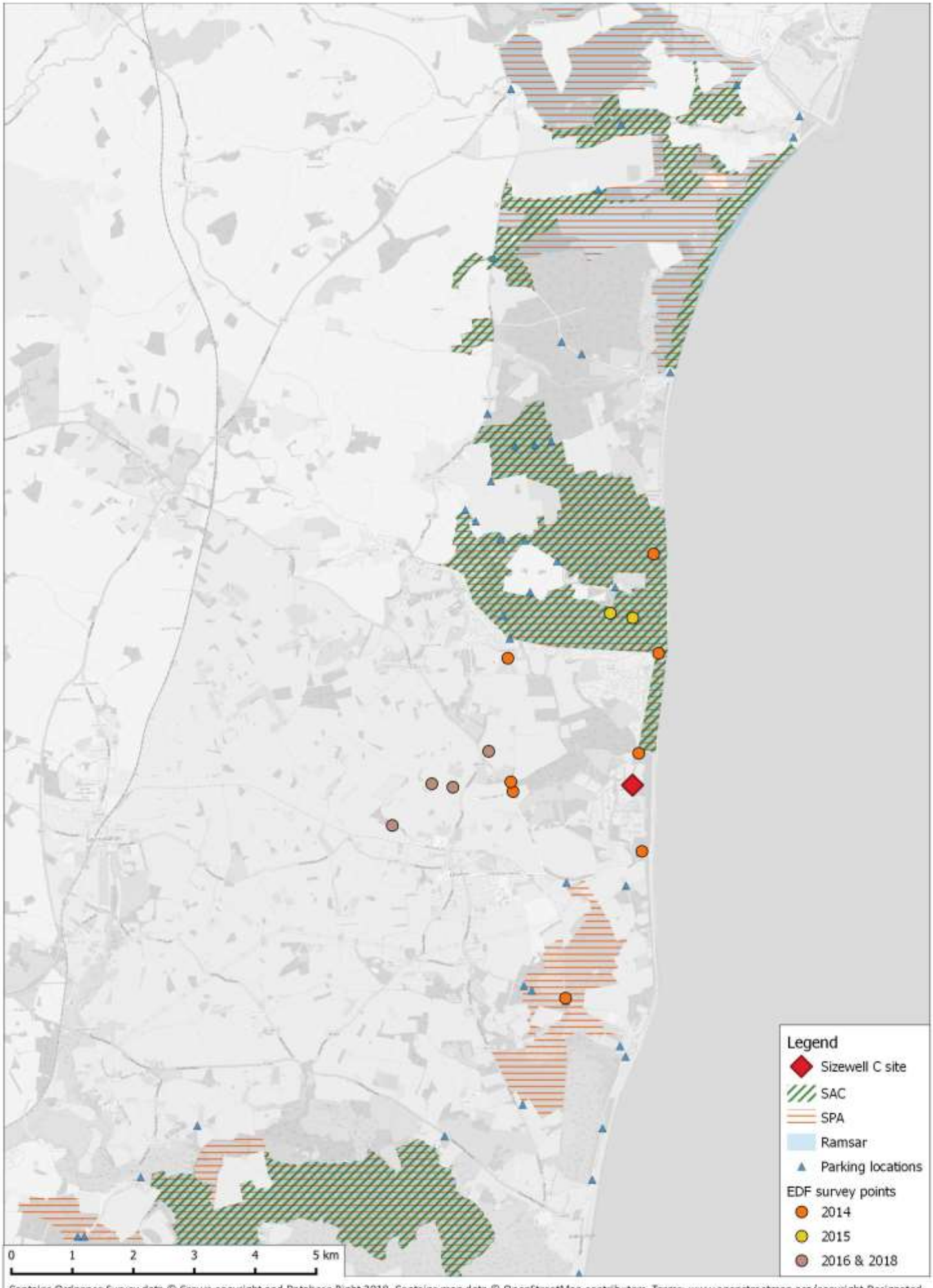
2.7 EDF commissioned visitor surveys to gain a more detailed understanding. These involved:

- 2014: 7 locations surveyed in August and November (514 questionnaires) summarised in 2014 **Sizewell C Visitor Surveys**;
- 2015: focussed on the Minsmere Reserve (133 questionnaires) summarised in **2015 Sizewell C RSPB Minsmere Visitor Surveys**;
- Additional surveys undertaken at 5 locations (surveyed in both August 2016 and November 2018, **2016-2018 Sizewell C Visitor Surveys**). These are however not referred to in the Shadow HRA recreational disturbance evidence base and are located away from European sites.

2.8 Map 1 details the locations for all the EDF surveys carried out in 2014, 2015, 2016 and 2018 in relation to the European sites. The map also shows a selection of parking locations around the European sites – locations where visitors might park when visiting them. These are shown as blue triangles and are not intended to be exhaustive. The map highlights that the survey coverage by EDF is minimal. In particular, there are no survey data from locations to the west of the Minsmere Reserve, around Westleton, or north of Dunwich Coastguards. Given the scant information in the existing baseline, we have little confidence that the survey effort is adequate to provide an understanding of visitor use and how that might change. We would have expected better spatial coverage. We would also have expected routes to be mapped and maps produced to show visitor numbers along different paths, allowing visitor data to be overlaid with ecological data. This is missing.

2.9 Surveys were conducted during August and November. While these months allow some comparison across the year, they are a poor choice. During August the coast is very busy with tourists and it might be expected that use by local residents would be reduced during this time. By contrast, November is likely to be more focussed towards local use, but likely to be quite limited. There is a notable gap of survey data from the spring. During the spring we would expect increased local use, making use of the warmer weather and longer days, with people coming from slightly further afield. Many countryside sites see marked peaks in visitors around the Easter period. The spring is also a time when breeding bird interest and other ecological interest is vulnerable to recreation impacts.

Map 1: Overview of European sites, parking provision, and EDF survey locations



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Predictions of changes in visitor use

Visits from the construction workforce

2.10 The **Shadow HRA recreational disturbance evidence base** (para 3.3.3) states that the approach taken to estimate recreational use by the construction workforce is based on the best available data and is precautionary. From a review of the data this does not seem to be the case.

2.11 EDF state in para 15.6.42 onwards of the **Amenity and Recreation Report** that the peak workforce would use different types of accommodation and provide the numbers in each:

- 7,900 workers;
  - 2,000 drawn from the existing population;
  - 5,900 non-home based ('NHB');
    - 880 buy homes;
    - 2,400 campus;
    - 600 caravans, and;
    - 2,000 renting (private rented sector or using tourist accomodation).

2.12 However, no evidence is given as to how these estimates are broken down and we suspect there is likely to be considerable guess work in estimating relative proportions. While these are 'peak' figures, it is clear that the workforce will number over 7,000 for 2 years (**Shadow HRA recreational disturbance evidence base** para 3.3.5).

2.13 We have constructed Figure 1 below to try and understand the numbers provided by EDF and understand their related estimates of countryside visits.

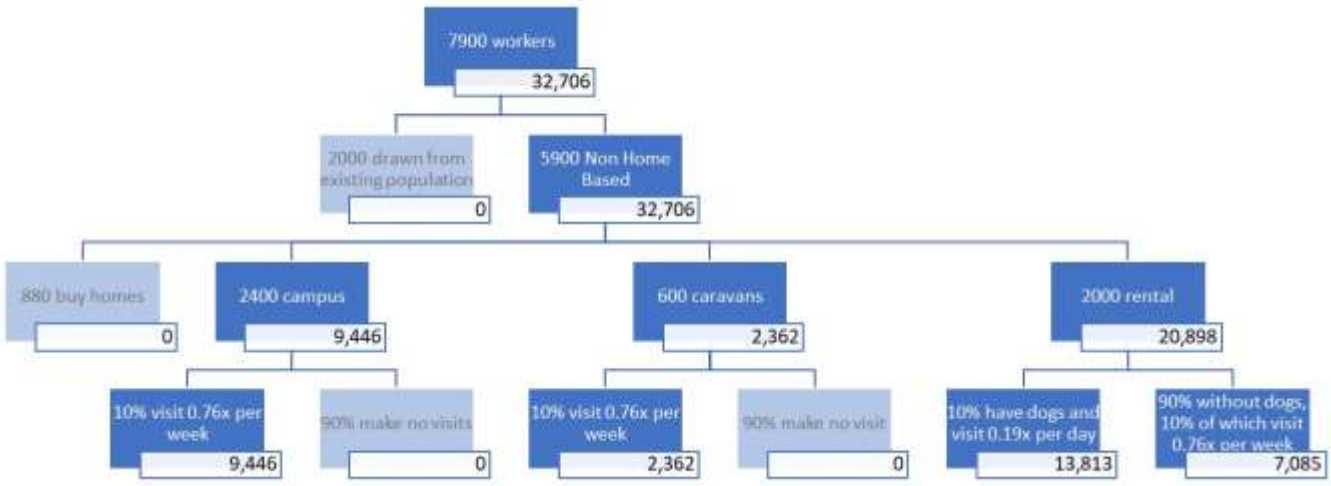


Figure 1: Chart summarising construction workforce and the estimates made of visits to the countryside by each segment. Pale boxes are those where no net increase in visits assumed by EDF. Numbers in bottom right indicate EDF estimates of countryside visits per year for each segment.

2.14 The 2,000 workers who are estimated to come from the existing population, and the 880 who buy their own homes, are discounted by EDF in terms of any recreational impact. EDF suggest they would not be net additional people to the area and therefore have no net additional effect on recreation activity (e.g. para 3.3.6). However, these workers should not be discounted, as they are more likely to visit the area around Sizewell C as they will be based there for work. Some of this workforce might travel independently to, and undertake recreational visits after, work – for example going fishing in the vicinity, walking the dog, or meeting friends or family when their shift has finished.

2.15 For the remaining 5,020 non home-based workers the assumption is made that 10% of those in rental accommodation (10% of 2,000 is 200) will have a dog or dogs and walk their pet daily. This 10% is apparently derived from a mystery shopper survey of rental providers to check how many properties allowed dogs. This rental shopper survey revealed marked variation and was very limited in scope. Many tourist lets, etc, may well have restrictions on pet ownership, but those workers living in rented accommodation are nevertheless considered more likely to own dogs than those living on the campus or in caravans (where dogs are not allowed) and as such are likely to seek out accommodation where pets are allowed. No consideration of this is made.

2.16 The number of dog walks is adjusted down to account for shift work (87%) and holidays (87%). It is not clear how the 87% figures have been derived and it would appear that the assumption is that dog walking would only take place on days when construction workers were doing shifts. The number of dog walks to local countryside sites is then reduced by a further 75% to account for 'short walks local to accommodation before and after shifts, often at night'. Again no justification is made for the application of the 75%.

2.17 For the rest of the 5,020 (i.e. those without dogs), 10% are assumed to undertake informal recreation in the countryside around Sizewell once a week (para 3.3.15 of the **Shadow HRA**). This appears to highly speculative, drawing on tourist board surveys from 2004 for visits by residents of Suffolk Coastal while discounting much more recent, national survey data (MENE) that gives typical visit rates to the countryside.

2.18 The resulting estimates for use of the countryside by the workforce are exceptionally low and far from precautionary. 7,900 workers are predicted to make 32,706 recreation visits to the countryside per year – this equates to around 4 visits per worker. The 32,706 visits are made up of 18,893 visits by those without dogs and 13,813 dog walks.

2.19 Whilst the 32,706 figure is given in the Shadow HRA, a very different estimate is confusingly given in the **Plants and Habitats Synthesis Report** which, on page 20, para 1.3.52, states that there will be 60,000 additional countryside visits per year from the construction workforce. In para 1.3.53 a total of 39,000 are anticipated for the key sites of Sizewell Beach/Sizewell Estate (20,000 annual visits), Aldeburgh (7,000 annual visits), Thorpeness (4,000 annual visits), Dunwich Heath (2,000 annual visits), Dunwich Beach (4,000 annual visits) and Minsmere (2,000 annual visits).

2.20 Further discrepancies are evident in the **Amenity and Recreation Report**. Table 15.7 gives a figure for 18,893 annual countryside visits made by workers without dogs. However, table 15.8 suggests construction workers with dogs would make 55,254 annual visits (after allowing for shift patterns and holidays).

2.21 EDF argue that the levels of recreational use of the countryside among construction workers will be low because: they will work long shifts; they may visit their family, etc/return to a permanent home at weekends; and they are likely to use gyms and sports facilities instead of the countryside, etc. These arguments are made with little justification or support. Market segmentation/demographic profiles produced by Sport England are referred

to (e.g. para 3.3.11 of the **Shadow HRA recreational disturbance evidence base**, see also the **Sport and Leisure Audit)** to justify assumptions about low levels of countryside recreation use. It is important to note that these profiles relate to organised sports, not countryside recreation use.

2.22 The **Shadow HRA recreational disturbance evidence base** totally fails to address that:

- The construction workforce will be based in a highly attractive area with lots of countryside recreation opportunities;
- The campus at Eastbridge is directly adjacent to numerous footpaths that provide access directly to the beach, woodland, heathland, and marshes – with multiple opportunities for activities such as mountain biking, jogging, fishing, photography, picnics, and walking;
- Some workers are likely to be drawn to work in the area and particular accommodation types due to recreation opportunities;
- Shift workers and those in temporary accommodation may use their weekends and non-shift time to explore the new area, and;
- There are increasing trends for informal recreational use of the countryside and that, following Covid, access patterns have changed markedly.

2.23 It is possible to drill into survey data and figures from other studies to highlight just how low the estimates from EDF are, with some consideration of relevant studies provided below.

2.24 Natural England's Monitoring Engagement with the Natural Environment (MENE) dataset provides national data on recreation use, with MENE data cited in the Shadow HRA recreational disturbance evidence base (para 2.4.18). Here, EDF's own report states that 3 in 5 adults in England make visits to the natural environment once a week and 51% of these visits were to the countryside or coast.

2.25 Drawing on the most recent MENE data (O'Neill, 2019), the average number of visits to the natural environment taken per person per week in 2018/19 was 1.7. This visit rate has been rising steadily (it was 1.3 in 2009). If this rate of 1.7 were applied to the entire workforce (7,900) across the whole year (52 weeks), this gives a figure of 698,360 visits per annum. Even if it is applied to just 45 weeks of the year, and to the 5,000 workers in private rental accommodation, and in caravans or on campus, the figure would be 382,500 visits to the natural environment. This figure is well over 10x as much as the 'precautionary' estimates made by EDF. While we concur this may still be an

overestimate, it does provide context as to how low the EDF estimates appear.

2.26 Sport England data (Gordon, et al., 2015) shows:

- 58% of the population 'Enjoy the outdoors';
- Out of the UK population of 43.7 million (aged over 16), some 32.4 million are 'active', i.e. take part in any kind of sport or outdoor activity, with 8.96 million (i.e. 20.5% of all those aged over 16) participants in outdoor activities;
- Males tend to be more active outdoors compared to females and activities such as mountain biking and running all have a male bias, and;
- 44% of gym users prefer to access outdoors, i.e. would prefer to be outside.

2.27 The Sports England Active Lives survey data can be accessed online with the online tool[1] and is able to provide information for particular sectors. The figures below are the percentage of people who walked for leisure/cycled (all cycling types) at least twice in the 28 days prior to the survey:

- National Statistics Socio-economic classes 6-8 (lower), i.e. those undertaking semi-routine occupations, routine occupations, or the long-term unemployed: 35.9%/12.7%;
- All adult males: 41.9%/21.1%;
- Those aged 16-34: 34.2%/17.9%;
- Those currently employed: 44%/19.4%, and;
- Single people: 36.8%/16.2%.

2.28 While these figures clearly do not single out those likely to be undertaking construction work at Sizewell C, they clearly indicate well over a third of individuals would likely walk for leisure (which includes dog walking) at least twice a month, and somewhere around 15-20% might be cycling (i.e. road cycling, mountain biking, etc). These are two activities which would be relevant to the construction workforce at Sizewell C.

Displaced visitors

2.29 The **2014 Sizewell C Visitor Surveys** (from 2014) indicate in para 4.1.17 that 65% of interviewees overall would not stop using the area around Sizewell C during construction, 29% would stop using it, and 2.5% said that they were not sure. Broadly about a third of visitors would choose to visit another

[1] https://activelives.sportengland.org/ accessed 06/08/2020

location, with the data suggesting that most would not be deflected very far and across a wide variety of locations.

2.30 The effect of displacement is estimated (in terms of additional visitor use) within the **Shadow HRA recreational disturbance evidence base (**paras 3.2.28 – 3.2.31**)**. The approach is fundamentally flawed. The screen shot below is taken from the **Shadow HRA recreational disturbance evidence base** (page 39 of the report), and, for example, calculates (erroneously) a figure of 4,380 visits to Aldeburgh. The errors in the calculation are set out below the screenshot. It should be noted that similar figures are used in other documents – for example the **Plants and Habitats Synthesis Report** (which gives a value of 4,377 for Aldeburgh in Table 1.5).

Table 3.5: Extrapolated visitor numbers for locations given by survey respondees as sites to which they would relocate during construction of SZC

			Most 'realistic' approach			'Precautionary' approach		
Column No.	1	2	3	4	5	6	7	8
Location	% of survey visitors displaced to location	Existing visitor numbers to location	Extra visits / year	Extra visits / day	% extra over existing visits	Extra visits / yr based on 500,000 visits in Sizewell area	Extra visits / day	% extra over existing visits
Aldeburgh	2.92%	150000	4380	12.0	2.9	14600	40	9.7
Butley	0.19%	10000	19	0.05	0.2	950	2.6	9.5

2.31 Below we use blue text to explain how the values in the table have been calculated by EDF and the black text describes why this is wrong, using the numbers given for Aldeburgh.

Column 1: 15 interviewees from the 2014 survey stated they would be displaced to Aldeburgh. 514 people were interviewed, giving the value of 2.92%. In the 2014 survey there were 15 interviewees who stated they would be displaced to Aldeburgh however 151 interviewees said they would be displaced somewhere and only 95 of those could name a location, presumably because they weren't sure. Therefore the % actually displaced to Aldeburgh could be higher.

Column 2: This is how many people EDF estimate visit Aldeburgh per year. The figure is essentially a rough guess, and the same value is applied to most sites. It highlights the poor quality of the data used.

Column 3: This figure is the 2.92% figure applied to column 2, i.e. 2.92% of 150,000. This step does not make sense and is where the fundamental error has crept in. It is not 2.92% of the people that visit Aldeburgh that are displaced, but 2.92% of the people visiting locations further north around Sizewell. The application of 2.92% to the number of visitors already visiting Aldeburgh makes no sense.

Column 4: This is simply 4380 divided by 365.

Column 5: This is simply column 1 repeated (and rounded down). This is the same as column 1 because the wrong numbers have been used in earlier columns.

Column 6: This is 2.92% of 500,000.

2.32 It would have been better to take the percentage of interviewees at each location where surveys were conducted that were displaced to different locations and then scale those figures up, based on the observation data at the relevant survey locations. In order to understand what that uplift might then mean, accurate visitor totals for the receptor sites is necessary and that is entirely missing from the EDF reports. The calculations given in columns 6-8 are more reasonable and make much more sense.

2.33 Within the 2014 visitor survey reports there is no breakdown of the percentages that would be displaced by survey point. This is critical information as it is necessary to understand the variation between locations.

2.34 The general approach of asking people whether they might be displaced and where they might go is useful, because there are few alternative means to ascertain such information. There are however concerns about such preference surveys and actual likely behaviour. This issue is discussed in the **Shadow HRA evidence base report** (para 3.2.5-3.2.12) but not really addressed. The text suggests some visitors may state they would be displaced but not actually change their behaviour, whereas no consideration is made that the opposite could occur. It is quite possible that some interviewees might not appreciate the scale of works and disruption and as such believe they wouldn't be displaced – but once work has started they may well be put off.

2.35 The **Shadow HRA evidence base report** (para 3.4.1) suggests that a precautionary approach is used and 29% of visitors are considered likely to be displaced. This is not precautionary, as it excludes those who indicated they weren't sure and does not address the concerns about preferences.

2.36 In summary, the figures for displacement are inaccurate, have been miscalculated, and the quality of the data in terms of the receptor sites and number of current visitors is very poor (meaning it is hard to be confident of what any uplift might mean). There has been a lack of survey work to accurately identify current visitor use around the area and current levels of use. The visitor surveys focus on a small number of locations but there are many more that have not been surveyed and for which no data exists.

Other visitor assumptions

2.37 There is no information provided to indicate how often and for how long the coast path will be closed and therefore how long the diversion is likely to be in place for. This is a key omission as the diversion will direct the coast path

and visitors through Eastbridge and across the Levels towards the Minsmere Sluice. This is a sensitive area in terms of disturbance and can be very wet and muddy, particularly during the winter. Displacement from here further north (e.g. when the path is waterlogged) would also deflect recreational use to sensitive locations. Without a clear understanding of how often the diversion will be required, and the duration and time of year it will be in place, it is impossible to assess the scale of impact.

2.38 Several assumptions are made within Tables 5.2 and 5.3 of the **Shadow HRA** concerning the behaviour of displaced site users. Item 9 in both tables suggests that new visitors to sites will exhibit the same behaviour and use the same access infrastructure as current site users, and that visitors specifically displaced from Sizewell and Minsmere to Aldeburgh would restrict their activities to the beach frontage around the town. Elsewhere in the same volume of the **Shadow HRA** (Para 8.3.64), it states that any recreational users displaced to Aldeburgh will focus their activities upon the main beach frontage, rather than spread out to nearby European Sites (e.g. the Alde-Ore Estuary). No evidence is provided to back up these assumptions.

2.39 The **Shadow HRA** also seeks to downplay the level of recreational activity occurring either side of the peak construction period, by contrasting it with that period (Para 7.4.73). Nevertheless, the period either side of peak construction will still have increased levels of recreational requirement above the current baseline. The same document also suggests that once construction has ceased many displaced site users will revert back to using Sizewell (Para 7.7.36) although no evidence is provided to support this, and it is considered just as likely that visitors will continue to use their newly adopted recreation sites post-construction.

Overall predictions of visitor numbers

2.40 Predictions of visitor use as a result of both the displacement of existing users and from additional construction workers are set out in the **Recreational Disturbance Assessment** Table 2.1. These should therefore be combined figures for both the construction workers and the displaced visitors. The numbers given bear little resemblance to those in other documents and it is not clear how the totals have been derived.

Screenshots of the top lines of different tables are provided on the following pages and all sites include estimates for Aldeburgh:

Recreational Disturbance Assessment. This claims to give figures for displaced recreational use and visits by construction workers – i.e. the overall increase in recreation use:

Table 2.1: Estimated additional visits to locations in the study area (two scenarios) as a result of displacement of existing users from the Sizewell area and the construction workforce.

Location and estimated number of annual visits	Access point(s) reference	% of survey visitors who gave as alternative location	Based on % of displaced visitors			Based on 500,000 visits to Sizewell area		
			Inc. visits / year	Inc. total visits / day	% inc. over existing visits	Inc. visits / year	Inc. total visits / day	% inc. over existing visits
Alde-Ore Estuary SPA								
Aldeburgh – 150,000	U,V,W	2.92%	9292	25.5	6.2	19,512	53.4	13
Butley – 10,000	DD	0.19%	19	0.05	0.2	950	2.6	9.5

Shadow HRA Recreational Disturbance Evidence Base. This table gives displacement of existing visitors

Table 3.5: Extrapolated visitor numbers for locations given by survey respondees as sites to which they would relocate during construction of SZC

			Most 'realistic' approach			'Precautionary' approach		
Column No.	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Location	**% of survey visitors displaced to location**	**Existing visitor numbers to location**	**Extra visits / year**	**Extra visits / day**	**% extra over existing visits**	**Extra visits / yr based on 500,000 visits in Sizewell area**	**Extra visits / day**	**% extra over existing visits**
Aldeburgh	2.92%	150000	4380	12.0	2.9	14600	40	9.7
Butley	0.19%	10000	19	0.05	0.2	950	2.6	9.5

Shadow HRA Recreational Disturbance Evidence Base. This table provides the displacement of construction workers

Table 3.9: Estimated numbers of construction worker visits to locations per year

Locations most likely to visit					Construction workers who have dogs
	*Estimated % who visit location	No. campus or caravan site based workers visits to location/yr**	No. PRS & tourist based worker visits to location/yr***	Total construction worker visits to location/yr	No. PRS & tourist based worker visits to location/yr****
Sizewell Estate and beach (not European Sites)	18%	2125	1275	3401	2486
Aldeburgh	26%	3070	1842	4912	3591
Thorpeness	15%	1771	1063	2834	2072

2.41 It can be seen that Table 2.1 of the **Recreational Disturbance Assessment** gives an increase of either 9,292 visits or a ('precautionary') 19,512. These are apparently displaced visitors and construction workers combined.

2.42 Table 3.5 of the **Shadow HRA Recreational Disturbance Evidence Base** gives totals for visitors that would be displaced: 4,380 and a ('precautionary') 14,600 per annum.

2.43 Table 3.9 of the **Shadow Recreational Disturbance Evidence Base** gives figures for use by construction workers: 8,503 per annum, this is the sum of 4,912 visits by non-dog walkers and 3,591 from dog walkers.

2.44 It can be seen that the figures do not add up. According to the EDF figures the total for Aldeburgh should be 23,103 (i.e.14,600 + 8,503) not the 19,512 referred to in Table 2.1 of the Recreational Disturbance Assessment. The errors are made for all sites, not just Aldeburgh. Even accepting our concerns regarding the approach, level of surveys etc., there are also different figures quoted in different reports with clear errors that undermine the conclusions drawn.

Impacts on nature conservation interest

2.45 Various locations ('amenity and recreation receptors') have been identified in the **Description of non-significant effects** report as experiencing non-significant effects. These include:

- Walberswick and Dingle Marshes;
- Middleton, Westleton and Darsham;
- South of Westleton, and;
- The areas around Thorpeness, Aldeburgh, and Knodishall.

2.46 EDF propose that for these locations, which include European sites, the effect on amenity (i.e. visitor experience) will not be diminished, i.e. for these locations the impacts from noise, use by construction workers, etc, will not affect visitor enjoyment.

2.47 The **Shadow HRA** appears to suggest in the early parts of the document (Table 2.2) that displacement of visitors is not one of the scenarios that is assessed and the only recreational effect listed is that from construction workers (at the bottom of the table).

Sandlings SPA

2.48 Table 5.3 (Item 9) within the **Shadow HRA** also suggests that any adverse effect arising from recreational disturbance upon Sandlings SPA is unlikely to impact SPA bird populations due to the extensive and spatially disparate nature of the habitats within the SPA. However, when considered in combination with the smaller size and number of parking spaces available within proximity to the SPA, it is considered that even a relatively small increase in visitor numbers could lead to a proportionately larger effect upon the fragmented habitat parcels comprising the SPA.

2.49 The percentage populations of Nightjar and Woodlark found within Sandlings SPA is presented as 3% and 9%, respectively, within the **Shadow HRA** (Para 6.3.198), although this has been calculated using the most recent 4-year average rather than the most recent (2018) counts. The recalculated percentages using the 2018 counts are 6% and 10%, respectively. Elsewhere in the document (Para 8.8.435) it is stated that Dunwich Forest, which lies outside of the Minsmere-Walberswick SPA but which supports a Nightjar population considered to be functionally linked with the SPA population, *has the capacity to accommodate a significant level of recreational use without any adverse effects on this species.* No evidence is provided to explain why this is considered to be the case.

2.50 Throughout the **Shadow HRA**, and the majority of supporting documents, recreational impacts upon Woodlark are only considered within the Sandlings SPA, despite the species being distributed across several localities (as evidenced by the desk study information provided within para 1.3.69 of the **Terrestrial Ecology and Ornithology Report – Ornithology Appendix**). Woodlark populations outside of Sandlings SPA (e.g. within Minsmere-Walberswick SPA) are considered likely to be functionally linked to the Sandlings SPA population and any impact upon them should therefore be considered.

Minsmere to Walberswick Heaths and Marshes SAC

2.51 **The Shadow HRA** argues that because there is already a high level of visitor use, any increase will not have an adverse effect. Para 7.7.34 states that there are already 1,114,206 visitors to the SAC. The paragraph then suggests that an additional 20,000 displaced visitors to the SAC would be small and the text also states "*this pressure would be diffuse and spread across a large*

number of potential car-park access points". There are many problems with this approach.

2.52 Firstly, the figure of 20,000 displaced visitors does not match that used elsewhere. For example, the **Recreational Disturbance Assessment**, in Table 2.1 on page 16, gives a total of displaced visitors of 23,172 or 88,623 visiting the Minsmere to Walberswick Heaths and Marshes SAC. These two totals are the sum of the increased visits per year given in the table and calculated by the two methods described.

2.53 To then argue that the increase is small in proportion to the total visitor numbers to the SAC, such that adverse effects can be ruled out, is flawed. Most visitors to the SAC visit the Minsmere RSPB Reserve car-park, Dunwich Cliffs, and other such destination car-parks that are promoted by the RSPB and National Trust. Wardens are present at these locations to meet and greet visitors, and considerable access infrastructure is already in place to manage recreation. The displaced visitors, and indeed the construction workforce, are likely to be pushed to inland car-parks and more informal parking around the periphery of the reserve (for example, across Dunwich Heath, around Westleton, or near Eastbridge). Here access is harder to manage as parking, etc, is not necessarily in the control of the conservation organisations and there are numerous footpaths and bridleways that provide access to areas with sensitive habitats.

Minsmere – Walberswick Heaths and Marshes SPA

2.54 Adverse effects are ruled out for Little Tern in the **Shadow HRA** (para 8.8.334) on the basis that management measures are in place to protect Little Terns (through the Little Tern Recovery Project) and subject to this management continuing. The Little Tern Recovery Project was funded through EU Life funds and funding is not secured indefinitely. It is therefore not a sound conclusion to rule out adverse effects on integrity on the basis that existing conservation management by organisations such as the RSPB will continue indefinitely. Given the highly mobile nature of the species, with colonies shifting over time, this assumption is even more open to question. Historically Little Terns have bred in front of Sizewell A and birds will exploit available habitat in areas with low levels of disturbance. Ensuring areas of beach with suitably low levels of recreation use is a challenge which will not be helped by the construction work and deflection of access.

2.55 Adverse effects on Hen Harrier are ruled out in the **Shadow HRA** (para 8.8.458), apparently because the roost sites are not known and confidential and because there is apparently ample habitat for birds to roost in within the SPA where no disturbance is likely. This is hardly a robust conclusion. Roost sites have been monitored for many years and have been in areas with public access. Locating roost sites would have been possible as part of the assessment (while ensuring they remained confidential) and more detailed consideration should have been undertaken.

2.56 Impacts upon Dartford Warbler are not considered in the EIA nor in any of the supporting ornithological survey reports. It is not a qualifying species within any of the SPAs, although this may potentially be seen as historical oversight in light of the important population now established in Suffolk (it is a SPA qualifying species elsewhere in its UK range). The species is also listed on Schedule 1 of the Wildlife and Countryside Act 1981 and is an Amber-listed Bird of Conservation Concern. Under these criteria alone the species should be considered alongside the other bird species identified in para 1.4.70 and Table 1.3 of the **Ornithology Synthesis Report**. It is likely to be present within areas of heath within which site users will potentially be displaced and the populations affected may be of value at the regional, or even national, level. Other non-SPA qualifying species (e.g. Stone-curlew) are considered within designated sites (e.g. Para 14.12.155 to 14.12.156 of the **Terrestrial Ecology and Ornithology Report**), and it is therefore not clear why Dartford Warbler has been excluded from assessment.

2.57 An overarching thread within the ornithological baseline surveys supporting the **Shadow HRA** is the age of the data. Table 1.7 in the **Terrestrial Ecology and Ornithology Report – Ornithology Appendix** provides the survey periods for the most recent ornithological surveys undertaken by Arcadis. Little Tern surveys have not been carried out since 2013, the last Nightjar surveys were in 2014, and breeding and wintering bird surveys have not been undertaken since 2015. Several historic species-specific ornithological surveys carried out in the period 2008 to 2012, and categorised as secondary data within the Ornithology Appendix, are also used to inform the assessment. It is therefore considered that update surveys would be required to accurately inform any impact assessment upon several of the important ornithological receptors identified.

3. The monitoring strategy proposed by EDF and adequacy

3.1 The **Shadow HRA** (e.g. para 7.7.37) states that "*The Rights of Way and Access Strategy for the EDF Energy estate would be developed to minimise the displacement of people away from the Sizewell C area and to nearby European sites to minimise trampling of vegetation. In addition, the strategy outlines a monitoring programme for recreational displacement to identify local mitigation measures, to be agreed with local land managers, which could be introduced to further reduce recreational disturbance*."

3.2 The above quote would appear to suggest that the strategy includes a monitoring programme. Elsewhere in the **Shadow HRA** the text regarding monitoring is more ambiguous and would suggest that a monitoring programme is yet to be produced. For example para 8.8.434: "*It is proposed that existing recreational management measures at Westleton Heath and other heathland areas within the Minsmere (southern) section of the Minsmere-Walberswick SPA, are enhanced to minimise the potential for any increase in recreational disturbance pressure on breeding nightjar and other breeding birds of heathland habitats. This would be a monitoring and mitigation plan, to be agreed with land managers and would be aligned with the proposed Rights of Way and Access Strategy*."

3.3 Further confusion results from para 8.8.111 which states: "*Mitigation measures have been identified to minimise the ecological effects of increased recreational pressure on habitats and associated species in the Minsmere RSPB Reserve area. A Rights of Way and Access Strategy is being developed to minimise the displacement of existing recreational users from Sizewell C and the likelihood that displaced visitors and construction workers would access the Minsmere-Walberswick SPA for recreation. In addition, the strategy outlines a monitoring programme for recreational displacement to identify local mitigation measures, to be agreed with local land managers, which could be introduced to further reduce recreational disturbance."* This seems to suggest that a strategy is being produced but is not yet available, and that the strategy includes the monitoring programme.

3.4 The **Rights of Way and Access Strategy** does exist; we have checked it for a monitoring programme and there is no programme set out. As such, it would appear that the monitoring of recreation issues has not been

programmed and that no monitoring strategy has so far been produced. It is therefore not possible to rely on monitoring as a means to deliver mitigation.

3.5 We would have expected more detailed visitor survey work as part of the assessment, with for example counts of parked cars at every parking location across the relevant European sites and supporting habitat. This would have provided basic baseline data on visitor numbers which would then provide the foundation for a monitoring strategy. Monitoring data could then be used to identify where changes occur and therefore where mitigation needs to be targeted.

4. The mitigation strategy proposed by EDF and its adequacy.

4.1 The Sizewell C Project would have an impact on various public rights of way (PRoW), including temporary and permanent closures and diversions. A **Rights of Way and Access Strategy** is proposed to resolve these issues. The **Rights of Way and Access Strategy** is referred to in the **Shadow HRA** (e.g. para 8.8.415) as the means by which mitigation will minimise the ecological effects of increased recreational pressure on habitats and associated species.

4.2 The Suffolk Coast Path, the future England Coast Path, and Sandlings Walk will need to be diverted inland for 'temporary periods'. The strategy does not give any indication of how frequently this will happen, the duration, etc. Without this information it is impossible to assess the adequacy of any mitigation to resolve disturbance impacts at Eastbridge and the surrounding vicinity.

4.3 The **Shadow HRA** is ambiguous about the state of mitigation proposals for recreation impacts. For example para 8.8.434: "*It is proposed that existing recreational management measures at Westleton Heath and other heathland areas within the Minsmere (southern) section of the Minsmere-Walberswick SPA, are enhanced to minimise the potential for any increase in recreational disturbance pressure on breeding nightjar and other breeding birds of heathland habitats. This would be a monitoring and mitigation plan, to be agreed with land managers and would be aligned with the proposed Rights of Way and Access Strategy*."

4.4 Exactly what the recreational management measures might be and how they would work to enable a conclusion that adverse effects on integrity can be ruled out is not clear. There appears to be scant detail on such mitigation and nothing in the **Rights of Way and Access Strategy**.

4.5 The **Shadow HRA** (para 8.8.438) seems to indicate that monitoring and mitigation will be linked and triggers have been set to deliver enhanced recreational management: "*mitigation in the form of enhanced recreational management measures is identified for heathland areas within the southern (Minsmere) block of the SPA during the construction period. If the changes in visitor use predicted during construction continue into the operational phase, then it is considered that these mitigation measures should be continued. Subject*

to the above enhanced recreational management, no adverse effects associated with increased recreational pressure are predicted on breeding nightjar using the Minsmere-Walberswick SPA, either directly or indirectly via habitat change."

4.6 It seems therefore that mitigation measures are being relied on and linked to monitoring, without the necessary detail provided to support the application and the conclusions in the **Shadow HRA**.

4.7 There are some mitigation measures described in the **Amenity and Recreation Report** (Section 15.5). These predominantly relate to retaining access provision, diversions to footpaths, and so on, rather than addressing impacts on habitats or species. Measures in the **Amenity and Recreation Report** that have some relevance include:

- A new off-road combined bridleway, cycleway, and footpath linking Sizewell and Gap to the construction accommodation campus (para 15.5.23-15.5.24);
- The provision of 27ha of new access land at Aldhurst Farm. The land here will include a small car park and informal and surfaced footpaths (para 15.5.26-15.5.28);
- Some enhancements to the Kenton Hills car park (para 15.5.25);
- Provision of formal sports facilities to the south of the Alde Valley Academy, including sports pitches (para 15.5.36), and;
- A footpath around the perimeter of the accommodation campus to allow workers to exercise within the campus boundary.

4.8 The **Mitigation Route Map** has been reviewed and does not provide any further detail on additional mitigation to the above that is relevant to recreation impacts upon European sites.

4.9 Additional (and conflicting) information on mitigation for recreational disturbance are provided in the **Terrestrial Ecology and Ornithology Report.** Para 14.12.161 states that it is considered prudent to develop appropriate mitigation for recreational disturbance issues. Para 14.12.162 states that Aldhurst Farm would provide 43ha of open space for informal recreation. In addition to this (para 14.12.163) indicates that local site-based measures "*would be developed as part of a Rights of Way and Access Strategy, in partnership with relevant stakeholders. This strategy would complement measures already outlined in the Suffolk Coast Recreational Disturbance Mitigation Strategy (RAMS) developed by Natural England and local planning authorities in Suffolk*". Such measures appear to be missing from the Rights of Way and Access Strategy.....

4.10 Further confusion arises in the **Planning Statement**, para 6.10.3 of which states: "*SZC Co. will monitor impacts on European Sites relating to recreational displacement during the construction phase as a result of the Sizewell C Project. SZC Co. will provide a European Sites access contingency fund to support measures to access points to European Sites where monitoring identifies a potential risk of harm occurring as a result of additional use of these areas. Measures to be funded could include additional signage and information boards, fencing, as well as training and support for wardens. The European Sites access contingency fund would not be available to support measures at RSPB Minsmere, or National Trust Dunwich Heath and Coastguard Cottages which will be funded through separate resilience funds (see section 6.14)*."

4.11 There appears to be no further detail on these measures. In order to be effective they must be timely, reliable, and achievable. There is no evidence given as to the level of funding that might be provided, what might trigger particular mitigation elements, and how they might be implemented. It is not really clear what training or support for wardens might look like – is this additional staff (and if so how will it be hosted) or simply a training course?

4.12 There is no detailed consideration or assessment as to how effective these measures might be, or how they will serve to allow a conclusion of no adverse effects on integrity to be made, in any of the reports. For example, we would expect to see information on the levels of use, types of visitors, and clear evidence that the mitigation is sufficient to address impacts from recreation use. In fact, the measures seem poorly conceived and inadequate to address the scale of impact that might occur. For example, new greenspace is provided in some parts of the country, such as the Thames Basin Heaths, at 8ha per 1,000 new residents, in order to resolve the issues from increased recreation associated with new development. While this metric is not necessarily transferable to the Suffolk Coast, it would suggest that – for the 7,900 construction workers alone – some 63ha of new greenspace might be necessary. Aldhurst Farm is merely 27ha and no evidence is given as to how it might function as new greenspace.

4.13 Furthermore, in the Thames Basin Heaths, and indeed many other parts of the country where there are new residents (from house building), on-site access management measures are provided alongside new greenspace. For example, increased wardening, fire-fighting resources, education, and awareness raising initiatives, and so on. These appear to be lacking for Sizewell C. There are many areas, such as around Eastbridge, from Eastbridge towards Westleton, and parts of Dunwich Heath and Dunwich

Forest where such measures could help address impacts from increased recreation.

5. Questions for National Trust/RSPB to raise in relation to recreation impacts.

5.1 Recreational disturbance impacts have not been properly assessed, evidence is lacking, and the suggested mitigation is confusing. Given the poor quality of the work done, it seems that a conclusion of no adverse effects on integrity cannot be justified. In some respects, we would advocate that EDF need to review, update, and redo much of the work on recreation impacts, with more original surveys necessary. Possible points to clarify/address include the following:

1. There are errors in the calculations of visitor estimates and discrepancies between the different reports. These need to be explained and clear analysis of the recreation impact issues and necessary mitigation presented. Without this information it is not possible to rule out adverse effects on integrity.

2. There is a lack of detail on assessment of ecological impacts. For example Little Tern, Hen Harrier, and trampling of shingle seem to be poorly addressed. For some of the coastal features there is little indication of how mobile species of dynamic habitats can be protected from disturbance. We would have expected to see much more detailed mapping of visitor flows and numbers (both current and predicted), providing heat maps of footfall. These maps would then be overlaid with ecological data and checked with site managers/local organisations[2].

3. Further information is required concerning the coast path diversion. How often will the diversion of visitors to Eastbridge be in place? How will parking issues be addressed in Eastbridge and how will the diverted path cope with increased use, given that it is prone to waterlogging in the winter?

4. There is a lack of mitigation to resolve recreation impacts at the following locations:

- Eastbridge area (including path to coast);

[2] We understand some checks were made early on in the assessment with relevant organisations, but the expertise of site managers in checking model predictions and pin-pointing where there are concerns with recreation impacts and how they should be addressed has been missed entirely.

- Periphery of Minsmere Reserve, particularly between Eastbridge and Westleton;
- Dunwich Heath;
- Beach at Dunwich, including vegetated shingle below the cliffs and marshes to north;
- Dunwich Forest;
- Walberswick Marshes;
- Beach area at Minsmere and south towards Sizewell;
- Beach area between Thorpeness and Aldeburgh (vegetated shingle);
- North Warren, and;
- Aldringham Walks.

5. As part of a robust mitigation package we would expect to see provision for an increased wardening presence, public education and awareness raising (including a focus on dog fouling and dogs running off-lead), protocols to resolve fire risks, etc. These seem to be alluded to in the **Planning Statement** but nowhere else.

6. There is no comprehensive monitoring programme to address visitor numbers and use. This needs to be drawn up and integrated into mitigation delivery.

6. Conclusions

6.1 We have reviewed the information provided by EDF in relation to visitor use and nature conservation impacts. This information is scattered across numerous reports and we have struggled to find a clear, coherent narrative that identifies the changes in recreation use might be expected were the proposal to go ahead, the scale of impact and the necessary mitigation that would be required.

6.2 The coastal strip in the vicinity of Sizewell is outstanding in its nature conservation interest and the area supports a range of habitats and species – within a relatively small geographic area – that is unique within the UK. The area is also a popular destination for recreation and draws visitors from the local area and more widely. There are visitor facilities and infrastructure in place at some key locations where, for example, tourist use is focussed. There are also numerous more informal locations where access is currently low key and difficult to manage. The Sizewell C proposal will change the recreation use through the increase in construction workers (some 7,900 workers at peak period) living locally and the displacement of existing visitors, for example as a result of footpath closures, diversions, traffic etc.

6.3 The area around Sizewell includes SAC, SPA and Ramsar sites and a range of qualifying features. There are a number of ways in which recreation impacts could undermine the conservation objectives for the relevant European sites. For example, through increased fire risk, disturbance, trampling, contamination and spread of invasive species. As such there are likely significant effects from recreation and appropriate assessment is necessary.

6.4 The purpose of the appropriate assessment is to undertake an objective scientific assessment of the implications for the European site's qualifying features potentially affected by the project in light of their conservation objectives. The competent authority should conclude that a project would not adversely affect the integrity of a European site, only if it is convinced that this is the case, such that there is no reasonable scientific doubt. In order to reach such a conclusion we would expect detailed survey work to show current levels of recreation, predictions of change and these overlaid with ecological data to show which locations (i.e. particular car-parks and paths) and which species/habitats would affected. Such a granular and detailed level of evidence has not been gathered. Were such information

available, it would allow mitigation measures and long-term monitoring to be designed and targeted to allow adverse effects on integrity to be ruled out.

6.5 Guidance on HRAs is clear that all mitigation measures should be effective, reliable, timely, guaranteed to be delivered and as long-term as they need to be to achieve their objectives (Tyldesley, Chapman, & Machin, 2020). It is for the competent authority, as part of the appropriate assessment, to assess the mitigation measures and the difference they would make to the anticipated project. The information provided by EDF does not provide the necessary level of precision in the baseline data or predictions of change. As a consequence, there is insufficient detail on the mitigation needed. Mitigation measures are not adequately set out or secured.

References

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O'Neill, R. (2019). *Monitor of Engagement with the Natural Environment – The national survey on people and the natural environment. Headline report 2019* (NECR No. 275). Natural England and the Office for National Statistics. Retrieved from Natural England and the Office for National Statistics website: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/828552/Monitor_Engagement_Natural_Environment_2018_2019_v2.pdf

Tyldesley, D., Chapman, C., & Machin, G. (2020). *The Habitats Regulations Handbook*. DTA Publications. Retrieved from https://www.dtapublications.co.uk/handbook/

Appendix: Other points/extra detail.

Note that footprint ref allows cross-reference between rows in this table only.

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
	Which document, by whom & reference	***Specific reference to locate the quote from next column – 'from' and 'to' if necessary***	***Copy/paste of the specific section that you have concerns with.***	***Explain why we are concerned with this section – either because it is not acceptable to us and we have a coherent challenge, or it is a significant risk to our position. We will not need to respond to everything we might disagree with.***	***A first draft of some next, notes or bullet points articulating how we might to respond to the issue or concern highlighted.***
1	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 4 – Section 2 - Para 2.1.7	*During the peak year of construction, it has been assumed that a total of 7,900 construction workers would work on the main development site at any one time and 600 construction workers would work on the associated development sites. The 7,900 construction workers at peak construction are assumed to be made up of:* *• 'home based' (2,016 workers); and* *• 'non-home based' (5,884 workers) comprising:* *− 2,400 workers on campus.* *− 600 workers in caravans.* *− 2,884 workers living off site.*	Use of the word assumption throughout, especially with respect to home workers, if used to underpin additional impact assessment.	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
2	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 4 – Section 2 - Para 2.1.8	*In addition, all 600 associated development workers are assumed to be 'home based'.*	Use of the word assumption. If not home-based then these workers will need to be accommodated elsewhere.	
3	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 27 – Table 2.2		Repeat of 'assumed' home worker numbers.	
4	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 84 – Table 5.2 – Item 9	*Additional visitors to these locations would be expected to follow similar behaviours to existing visitors and use the defined path network/beach. Further incursion by people into areas supporting sensitive shingle vegetation would not be expected and additional loss of vegetation as a result of trampling is therefore considered unlikely to arise.*	No indication as to why novel/increased numbers of visitors would be expected to follow the same pattern.	
5	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 84 – Table 5.2 – Item 9	*It is considered that the majority of additional visits undertaken by people displaced from Sizewell, or potentially the RSPB Minsmere Reserve, to Aldeburgh would involve activities on the immediate beach frontage around the town, rather than the estuarine habitats and landscape of the Alde-Ore Estuary Ramsar site.*	No indication as to why people visiting Minsmere (a "wild" site) would swap to the town seafront if displaced. Different pull factors at work potentially.	
6	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 93 – Table 5.3 – Item 9	*Many of these areas, in particular the coastal frontage/beach at Dunwich and Walberswick, are already subject to intense visitor use.*	Leading statement which suggests that no cumulative ecological effect can occur at a heavily used site with	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
				the advent of increased visitor numbers.	
7	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 94 – Table 5.3 – Item 9	Same wording to Footprint ref 5	No indication as to why people visiting Minsmere (a "wild" site) would swap to the town seafront if displaced. Different pull factors at work potentially.	
8	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 94 – Table 5.3 – Item 9	*Such increased recreational pressure is likely to involve activities on the immediate beach frontage at Aldborough, rather than on the estuarine habitats of the Alde-Ore Estuary SPA*	No indication given as to why this would be the case.	
9	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 94 – Table 5.3 – Item 9	*However, despite all this, the potential for an increase in the number of visits to some locations around the Alde-Ore Estuary cannot be discounted and further studies need to be carried out to remove this pathway of effect.*	Cannot rule out disturbance effects on the SPA.	
10	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 95 – Table 5.3 – Item 9	*Sandlings SPA includes a number of discrete areas of heathland, acid grassland and coniferous plantations. Within the 8km displacement buffer zone the SPA component area comprises Aldringham Walks, to the south of Sizewell, and North Warren Nature Reserve located to the west of Thorpeness and north of Aldeburgh. Both of these sites are located away from the immediate coast and are more discrete than the sites at Minsmere and Dunwich. As a result, they have fewer visitor facilities and are accessed via a series*	No explanation of 8km displacement buffer. Suggestion that sites won't be visited because they are not coastal. Smaller size/smaller amount of parking suggests potential for increased impacts arising from relatively	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			of smaller and informal car parks. Given the extensive and spatially disparate nature of the habitat comprising the SPA is that an adverse effect from Sizewell C at the level of designated SPA populations is considered to be unlikely.	small increase in visitor numbers.	
11	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 125 – Section 6 - Para 6.3.3	*The citation and most recently available population estimates for the SPA show declines in numbers for the breeding qualifying features, other than marsh harrier, whilst recent population estimates for the wintering populations are higher than at citation (Table 6.1). Declines are particularly notable amongst the breeding seabirds, with neither of the two tern species which are qualifying features of the SPA currently being present as breeding species.*	Phrasing here is leading, intimating that quality of designated site has declined. No indication as to why this may be (e.g. increased recreation) or the issues posed to recovery by increased visitor numbers arising from Sizewell C project.	
12	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 127 – Section 6 - Para 6.3.11	Said of Little Tern: *The peak count was eight birds at VP10 (Sudbourne Beach) in early June 2013, at which point a colony attempted to establish. However, this was deserted by the time of the nest survey in late June.*	No indication given as to why colony abandoned. If due to recreation then evidence of potential impact on re-colonising feature of SPA.	
13	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 127 – Section 6 - Para 6.3.13	Said of Little Tern: *Based on the mean maximum alongshore foraging range (3.9km) suggested by Parsons et al. (Ref. 6.43), breeding birds from a colony at Slaughden could be expected to generally forage no further south than Orford Ness and no further north than Thorpeness.*	Mean value is being misinterpreted here. Could easily have birds regularly foraging at distances past this point.	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
14	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 143 – Section 6 - Para 6.3.48	*Given that the mean maximum foraging distance from breeding colonies in the SPA is expected to be around 3.9km alongshore (Ref. 6.43), the nearest VP is likely to be outside the maximum foraging range for little terns in this SPA. However, the greater distance of the Benacre to Easton Bavents SPA little tern colony from the main development site also means that there would be less potential for effects to arise on this SPA little tern population than on those at SPAs closer to the main development site.*	Same misinterpretation of mean foraging range (see Footprint ref 13).	
15	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 143 – Section 6 - Para 6.3.49	*Based on the mean maximum alongshore foraging range (3.9 km) estimated by Parsons et al. (2015 – Ref. 6.43), breeding birds from the colony at Covehithe Broads could be expected to generally forage no further south than Southwold and no further north than Kessingland.*	Same misinterpretation of mean foraging range (see Footprint ref 13).	
16	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 145 – Section 6 - Para 6.3.54	*However, there have been marked declines in the breeding populations of little tern, teal and nightjar, and in the wintering population of hen harrier.*	No indication of why these decreases have occurred. Potentially due (at least partly) to increased levels of recreational pressure/disturbance?	
17	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 148-149 – Section 6 - Para 6.3.66	*Although these survey data are relatively old, there has been relatively little change in the relevant habitats in the Minsmere South Levels and Sizewell Marshes and no substantive increase in bittern numbers within the RSPB Minsmere Reserve since*	Stated change in habitat use. No way of confirming that behaviour and distribution hasn't	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			these surveys. Therefore, it is likely that these findings remain representative of the current situation.	changed without update surveys.	
18	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 149 – Section 6 - Para 6.3.67	*Other evidence on the use of the Sizewell Marshes by bittern derives from radio-tracking studies undertaken by RSPB in 2000 and 2001, which showed that the Sizewell Marshes were used by first winter birds from the SPA (Ref. 6.73).*	20 year old data from a population which was just starting to recovery. Behaviour/movements potentially different now due to increased population/distribution of territories on site.	
19	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 168 – Section 6 - Para 6.3.107	*The Minsmere South Levels and Sizewell Marshes (which occur outside the SPA – Figure 6.2) comprise habitat that may be used by breeding gadwall, so there is the potential for functional linkage with the SPA gadwall population.*	Misleading wording. Following paragraph makes it clear that both localities are used by breeding Gadwall.	
20	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 168 – Section 6 - Para 6.3.110	*The Minsmere South Levels and Sizewell Marshes (which occur outside the SPA – Figure 6.2) comprise habitat that may be used by breeding shoveler, so there is the potential for functional linkage with the SPA shoveler population.*	Misleading wording. Following paragraph makes it clear that both localities are used by breeding Shoveler.	
21	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 170 – Section 6 - Para 6.3.115	*Thus, surveys of the Kenton Hills and Goose Hill plantations, undertaken specifically for nightjar, recorded no birds in any of the three visits (i.e. 19th May and 23rd June 2014 and 17th May 2015).*	Are 3 visits enough to confirm probable absence? Also these surveys are 5 years old…	
22	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 171 – Section 6 - Para 6.3.118	*Two surveys (each of three hours duration) were undertaken on each survey day, with two hen harrier sightings obtained over the wintering period (in October and November survey) and both involving*	Evidence of Hen Harriers favouring South Levels (for foraging at least).	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			birds hunting over the Minsmere South Levels (Ref. 6.45)... Four of the records were from the Minsmere South Levels and six from the Sizewell Marshes.		
23	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 182 – Section 6 - Para 6.3.154	*Greater numbers of shoveler were recorded during WeBS counts within the WeBS Sizewell Belts count sector (with peak counts of 10 or more birds in six of the seven years for which data were obtained) than during project-specific surveys of the Sizewell Marshes, which recorded only two birds in this area during the two winters of survey (Table 6.14 and Table 6.15). Such a marked discrepancy between the respective counts is difficult to explain, although it is notable that shoveler were recorded in a relatively low proportion (i.e. 60%) of the WeBS Sizewell Belts counts.*	Potentially evidence of too low a survey effort in project-specific surveys?	
24	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 203 – Section 6 - Para 6.3.197	*Although the SPA is within 0.7km of the main development site at its closest point, the largest blocks of land which comprise the SPA are considerably further from the main development site, at over 9km.*	Still a material consideration though!	
25	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 203 – Section 6 - Para 6.3.198	*Over this period the number of nightjar territories on the Reserve increased from two in 2010 to five in 2018 (with an average of 1.4 territories per year from 2010 to 2014 and of 2.6 from 2014 to 2018), whilst for woodlark they increased from three in 2010 to seven in 2018 (with an average of 4.6 territories per year from 2010 to 2014 and of 6.8 from 2014 to 2018) (RSPB, unpublished data). Therefore, the discrete block of the SPA which is closest to the main*	The author is being disingenuous and using the most recent 4-year average rather than the most recent (2018) counts to calculate percentage of population. Using most recent counts the	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			development site seems likely to hold approximately 3% and 9% of the total SPA nightjar and woodlark populations, respectively.	percentages come out at 6% and 10% for Nightjar and Woodlark, respectively.	
26	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 258 to 259 – Section 7 - Para 7.4.73	*There is the potential for an increase in visitor numbers or changes in the patterns of use of recreational areas (such as the Alde-Ore Estuary Ramsar site) due to the presence of the Sizewell C Project construction workers. For the purposes of assessment, it is assumed that 7,900 workers would be present for the peak construction period of the Sizewell C Project. They are likely to be on-site for approximately 20% of the overall construction timeline (2.5 years of the 9 to 12 year construction period) and increase the pressure associated with recreational activity. Either side of peak construction the workforce would be lower and, therefore, would generate less demand for informal recreation.*	Assumptions of worker distribution again, and suggestion that either side of peak construction that recreational pressure will still be higher than it is now.	
27	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 259 – Section 7 - Para 7.4.74	*In addition, the construction period could lead to the displacement of a number of existing recreational users through their avoidance of particular areas, e.g. Sizewell Beach, during the construction period. When considered together, these effects are expected to generate an additional 60,000 recreational visits per annum overall, of which 7,000 may be to Aldeburgh, where it is possible to access the Alde-Ore Estuary Ramsar site.*	No indication of where these figures have come from.	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
28	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 259 – Section 7 - Para 7.4.76	*Appendix E of this report indicates that the car park locations that give access to the Ramsar site already receive an estimated 580,000 recreational visits per year, and that any increase due to recreational users displaced from Sizewell would be small (estimated to be an additional 29,000 recreational visits in this case per annum). In addition, this total increase in pressure would be diffuse and spread across a large number of potential car park access points.*	Previously stated that displacement will lead to use of existing heavily used sites – not sure that this fits with diffuse? The numbers stated still comprise a 5% increase in visitor numbers.	
29	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 259 to 260 – Section 7 - Para 7.4.79	*The number of workers for the operation phase would represent a substantial reduction from the levels during construction. The assessment for recreational disturbance during construction concluded that adverse effects would not arise and, as less recreational visits would be generated, it can be concluded that recreational disturbance during the operation of the Sizewell C Project would not have an adverse effect on the vegetated shingle of the Alde-Ore Estuary Ramsar site.*	The workers are still an increase in "population" above current levels, just fewer than in the construction phase…	
30	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 273 – Section 7 - Para 7.7.31	*The Plants and Habitats Synthesis Report as provided in Volume 2, Chapter 14, Appendix 14B1 concludes that any trampling effects are likely to be reversible in time, with vegetation recovering once the trampling pressure is removed.*	Assumption that trampling pressure will be removed and that people's behaviour doesn't change as a result of displacement during construction. Also no confirmation that trampling effects are reversible in reality,	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
				if damage to underlying seedbank/invasive spread/eutrophication occurs as an allied impact. All of this is subsequently highlighted in the remainder of the same paragraph.	
31	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 273 to 274 – Section 7 - Para 7.7.34	*In addition, this pressure would be diffuse and spread across a large number of potential car-park access points. It is estimated that perhaps 20% of the predicted increase in visitor numbers would visit the main RSPB nature reserve (where dog walking is not permitted) and 80% of the predicted increase in visitor numbers would visit the outer areas of the reserve, where dog walking is permitted.*	It's not clear in the shadow HRA how these visits will be spread across numerous car parks, or how this spread has been calculated. Likely to be a few busier and many quieter sites.	
32	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 274 – Section 7 - Para 7.7.35	*There is no automatic correlation between an increase in the number of recreational visits and the potential for the qualifying features of European sites to be detrimentally affected.*	It's arguable that published evidence suggests there is a correlation with "potential for" (even if impacts do not ultimately occur).	
33	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 274 – Section 7 - Para 7.7.35	*It is envisaged that, overall, managed sites with a well-defined path network, such as the core area of the RSPB reserve, where people are easily observed, new recreational users are likely to keep to existing*	How is this justified? Is there any evidence to prove that this is in fact the case?	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			path networks and would be unlikely to lead to an increase in trampling.		
34	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 274 – Section 7 - Para 7.7.36	*The displacement of existing recreational users is likely to last for the duration of the construction phase, although as people establish new patterns of behaviour and access alternate sites, the changes could become permanent for a proportion of users. For the majority, once construction activities have ceased, a substantial proportion of displaced users are likely to reuse the Sizewell area and be joined by new users.*	No evidence is provided to suggest how likely any of these scenarios are. Increased recreational pressure could remain indefinitely.	
35	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 274 – Section 7 - Para 7.7.37	*As set out earlier, mitigation measures have been established to minimise the requirement for both construction workers and existing recreational users from Sizewell to access the SAC for recreation.*	Assume this refers to the Suffolk RAMS (alongside proposed Rights of Way and Access Strategy for the EDF Energy estate?	
36	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 285 to 290– Section 7 – Table 7.8	Table cumulatively lists 1,100 new dwellings in outline planning applications within 13.7km of Sizewell C, and a minimum of 815 new dwelling allocations within various relevant Plans within 8.6km.	Table identifies potential for in-combination effect with Sizewell C, but not for adverse in-combination effect. This is presumably due to existing RAMS Strategy (mitigation??)	
37	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 301 to 306 – Section 7 – Table 7.10	See Footprint ref 37	See Footprint ref 36.	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
38	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 344 – Section 8 – Para 8.3.64	*Aldeburgh was identified as the most likely site near to the Alde-Ore Estuary SPA that users could be displaced to, with smaller predicted increases to other locations. Increased visitor use in and around Aldeburgh would be unlikely to impinge directly upon the estuary, with the focus of any additional activity likely to be the main beach frontage.*	No evidence provided here to confirm limitation to Aldeburgh beachfront.	
39	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 346 – Section 8 – Para 8.3.73	*Any residual effects of visitor displacement from Sizewell C post-construction, and recreational use of the local area by operational workers, are predicted to be so small as to be undetectable in the context of the likely future drivers of trends in recreational use of the Suffolk Coast (population growth, initiatives to promote tourism and the increasing popularity of outdoor recreation) and the predicted numbers of annual visitor trips to the Suffolk Coast in the vicinity of the main development site.*	Is this a material consideration when assessing the impact of the Sizewell Project?	
40	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 494 – Section 8 – Para 8.8.422	*It is also unlikely that SPA birds rely on suitable habitat near to the main development site for foraging because the main breeding sites within the SPA are over 1km from the main development site. Although nightjar have been recorded foraging at distances of up to 3.1km from their nests, most foraging is generally much closer to the nest and studies of radio-tracked birds in southeast England recorded a mean maximum distance of 747m from the territory centres*	Would argue that it is less likely, but not unlikely.	

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41	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 496 – Section 8 – Para 8.8.434	*It is proposed that existing recreational management measures at Westleton Heath and other heathland areas within the Minsmere (southern) section of the Minsmere-Walberswick SPA, are enhanced to minimise the potential for any increase in recreational disturbance pressure on breeding nightjar and other breeding birds of heathland habitats. This would be a monitoring and mitigation plan, to be agreed with land managers and would be aligned with the proposed Rights of Way and Access Strategy.*	Not currently clear what this would entail.	
42	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 496 – Section 8 – Para 8.8.435	*It is considered that Dunwich Forest – outside the SPA but with a functionally linked breeding population of nightjar - has the capacity to accommodate a significant level of recreational use without any adverse effects on this species.*	Why? No evidence for this statement provided.	
43	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 565 – Section 8 – Para 8.11.23	*Estimated increases in visitor numbers to locations associated with the SPA are shown in Table 8.31. The largest estimated increase is at Thorpeness Village. Most visitors relocating here might be expected to access the beach and village. Although there is access to the Sandlings SPA to the west (Aldringham Walks) this is a 4-5km round trip.*	Confirmation that people will not access areas away from the beach?	
44	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 566 – Section 8 – Para 8.11.26	Said of Nightjar*: It is also stated that the population size at classification may have been unusually high due to the extensive area of open habitat suitable for nesting that was created following a gale in 1987 that felled most trees; and a more realistic target is being modelled.*	If this is the case, then it increases the importance/proportion of the population within those parcels closest to the Sizewell C site.	

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45	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 566 – Section 8 – Para 8.11.26	*Disturbance due to human activity is identified as a potential issue for nightjar if an increase in development locally leads to an increase in recreational pressure (Ref. 6.109), suggesting it is not currently considered to be a limiting factor.*	This interpretation is open to question.	
46	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 566 – Section 8 – Para 8.11.27	*The Sizewell C visitor survey data and presence of several entry points to open access land indicate that the Sandlings SPA is already subject to considerable visitor use. There are extensive path networks and access to woodland and open habitats. The main activity undertaken by visitors is dog-walking (Ref. 8.139). Nevertheless, as described above RSPB data show that the nightjar population within the RSPB's Aldringham Walks and North Warren reserve has increased in recent years*.	Not clear that visitor survey data shows high levels of use within Aldringham Walks and North Warren Reserve, specifically?	
47	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 566 – Section 8 – Para 8.11.28	*No change in the behaviour of users or the pattern of use is considered likely to arise.*	Based on what data? Increased site use may cause existing users to spread out more within the site.	
48	5.10 Shadow Habitats Regulations Assessment Volume 1: Screening and Appropriate Assessment 1 of 5 (revision 1.0)	Page 568 – Section 8 – Para 8.11.36	*Subject to the implementation of enhanced management measures for recreational access at Aldringham Walks and North Warren, an increase in recreational pressure during the construction of Sizewell C is not predicted to adversely affect the ability of the Sandlings SPA to achieve the conservation objectives for breeding nightjar, either directly or indirectly, via effects on habitats.*	How will these be enforced? Increasing population is a sign of recovery and increased disturbance should therefore be avoided.	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
49	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 46 – Para 14.7.36	*The Plants and Habitats Synthesis Report (Appendix 14B1 of this volume) concludes that any trampling effects are likely to be reversible with vegetation recovering once the trampling pressure is removed*	Assumption of non-continuation of use at end of 10-year period or change in structure in the interim.	
50	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 47 – Para 14.7.44	*Once construction activities have ceased, a substantial proportion of displaced users would likely reuse the Sizewell area and be joined by new users.*	Assumption.	
51	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 59 – Para 14.7.108	*In addition, this total increase in pressure would be diffuse and spread across a large number of potential car-park access points.*	Evidence for this?	
52	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 157 to 162 – Table 14.23	No mention of Dartford Warbler anywhere, despite other non-qualifying species (with respect to designated sites) included in table.	Unsure why this species is absent. Presumably present on heaths within ZoI and of importance at a county (regional/national?) level and prone to disturbance.	
53	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 170 to 177 – Table 14.25	No recreational disturbance effect indicated for Hobby	Potential disturbance effects near nest sites, where they occur following displacement of visitors	
54	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 208 – Para 14.12.154	*For example, a well-used site, with wide, clearly defined access tracks, in which visitors behave in a similar manner and remain on the path network,*	What about species like Dartford Warbler, etc, where an increase in	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			could potentially have the capacity to absorb many additional visits.	passes along paths has been shown to affect breeding success?	
55	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 208 – Para 14.12.155	*For example, the RSPB has indicated that disturbance to the core RSPB Minsmere Reserve is unlikely as dogs are not allowed and access is managed, but the outlying heath and grassland areas are more vulnerable to recreational disturbance and in particular ground nesting species such as stone curlew. An increase in recreational disturbance is likely to last for the duration of the construction phase. In the absence of mitigation, a significant effect could occur.*	Still no mention of Dartford Warbler, although Stone Curlew (a non-qualifying species) gets a namecheck.	
56	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Page 211 – Para 14.12.168 – 5th bullet	*Recreational pressure. Once the proposed development is operational, access to the EDF Energy estate and the beach would return to conditions similar to the existing situation and no displacement of recreational users is expected.*	No mention of potential permanent increase in visitor levels at novel sites post-construction, either through retained practise or/and increased population (workers stay on scenario).	
57	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	Throughout		Entire report only talks about Woodlark breeding within Sandlings SPA. Are they not present elsewhere (e.g. Westleton) and	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
				comprise supporting populations?	
58	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14B2: Ornithology Synthesis Report	Page 50 – Para 1.4.137	No mention of Dartford Warbler, or of Woodlark away from Sandlings SPA, with respect to recreational disturbance		
59	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14B2: Ornithology Synthesis Report	Page 52 – Para 1.4.142	*Little terns have historically nested at Sudbourne Beach, near Slaughden. Sudbourne Beach is potentially accessible via a car park south of the village of Aldeburgh. The Recreational Evidence Base concludes that any increase in visitor numbers as a result of potential displacement from Sizewell C (estimated at 2.9 - 13% for this location) would not increase the disturbance pressure at this site. This takes account of existing access management measures, including prohibition of dogs on the beach between 1 May and 30 September, signage, fencing and vehicle barriers.*	No indication as to why no increased impact assumed upon Little Terns.	
60	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14A7: Ornithology Part 1 of 2	Page 18 – Table 1.7 – Breeding bird surveys row		Only single monthly visits carried out, with 2 in total in 2015.	
61	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14A7: Ornithology Part 1 of 2	Page 37 – para 1.3.69	*Within the wider landscape, woodlark have been reported from "Sizewell" in the desk-study (likely to be within the survey area), RSPB Minsmere Reserve, Eastbridge, Aldringham Common, Thorpeness, Leiston and Sizewell Common. RSPB reported woodlark present within RSPB Minsmere Reserve*	First mention I've found anywhere of Woodlark outside of Sandling SPA. Impacts of recreation don't appear to have been considered in these other locations.	

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			(peak count of 22 birds) and RSPB North Warren Reserve (peak count of 53 birds) (Ref. 1.32). RSPB also provided data for 2014 and 2015. A total of 34 woodlark records were recorded in RSPB Minsmere Reserve, with a peak count of 21 pairs in 2015. Other records of woodlark were from within the wider Minsmere to Walberswick Heaths and Marshes SPA at Dingle Marshes, where four records were reported, a peak count of nine pairs was recorded in 2014, and Snape, where six records were reported with a peak count of 10 pairs.		
62	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14A7: Ornithology Part 1 of 2	Page 37 – para 1.3.74	*No woodlark was recorded by Arcadis during either the breeding bird surveys or the Winter bird surveys between 2014 and 2015.*	Evidence of Arcadis surveyors missing species during their surveys, or of reduced survey effort?	
63	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology – Appendix 14A7: Ornithology Part 1 of 2	Appendix – Sizewell – First Interim Bird Report - Feb 2008 – page 25 (page 210 of pdf)	*Surveys for woodlark are best carried out between February and June, and if a three visit strategy is adopted as was undertaken for the national survey in 1997 (Wotton, 1997), only one territory mapping visit would fall within the survey period that this report covers (April to July inclusive).*	Indication that Woodlark survey effort not within optimum period to record the species.	
64	EN010012-001884-SZC_Bk6_ES_V2_Ch15_Amenity_and_Recreation_Appx15A_15J_Part_1_of_3	Appendix A, page 6 (page 7 of pdf), para 2.1.2	Research and field based questionnaires identified that the approximate median1 distance likely to be travelled by people to reach a location for recreational activities is 8km. This was defined through review of selected published reports on recreational visitor surveys in landscapes of a	Use of term buffer zone and receptor zone is confusing. Not clear why the median is used. Would imply half of use not accounted for.	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
			similar type to that within the area around the Sizewell C site, in particular either heathland landscapes or estuary landscapes associated with European habitat designations, including SPA and SAC designations. The evidence base for this is set out in the Sizewell C Proposed Nuclear Development HRA Evidence Plan Volume II (of II), Appendix 3.5 SZC-EP-W4-002 Disturbance due to potential increase in recreational pressure (EDF Energy October 2014). As such, this zone is judged to be the appropriate extent of the catchment area for visitors that have the potential to be displaced by changes to PRoW and access areas within the Indicative Main Development Site during construction and operation of Sizewell C. The Displacement Zone captures a number of settlements which have been used to define the Buffer Zone (see below).	Furthermore, if someone lives 8km away and chooses to visit somewhere else instead of Sizewell, then that means they might visit somewhere 16km away from the power station, not 8....	
65	EN010012-001768-SZC_Bk5_5.10_V1_Shadow_HRA_Report_Part_4_of_5	Appendix E: Recreational Disturbance Assessment page 19 (page 324 of pdf) para 3.2.9	Dolman (2010) (Ref. 3.12) studied recreational disturbance and predation, and the potential for interaction between these two factors, on breeding nightjar and woodlark. The study site was Thetford Forest, part of the Breckland SPA in Norfolk and Suffolk.	Para fails to point out that the visitor use of Breckland sites is very different (much lower) than Suffolk coastal sites. Same issue relevant to para 3.2.15	
66	EN010012-001768-SZC_Bk5_5.10_V1_Shadow_HRA_Report_Part_4_of_5	Appendix E: Recreational Disturbance Assessment	For example, a well-used site, with wide, clearly defined access tracks, in which visitors tend to behave in a similar	Implies birds will adapt to disturbance and become habituated. Evidence that birds can	

Footprint ref	DCO Document name & reference	Page, Section, Paragraph reference	Quote of text from document	Issue or concern	Response
		page 27 (page 332 of pdf) para 3.2.9	manner (such as remaining on paths) could potentially have the capacity to absorb significant numbers of additional visits. In such situations, the existing ecological interests may be adapted and habituated to this pattern of recreational usage and significant increases....	habituate is scant and as for adaptation...	
67	6.3 Volume 2 Main Development Site Chapter 14 Terrestrial Ecology and Ornithology Appendix 14B1 Plants and Habitats Synthesis Report APP DOC 250	Para 1.3.49, page 17 (Page 20 of pdf)	States use of 500,000 figure is precautionary as an estimate of visitor use. " This approach is considered to be precautionary in that it assumes that visitor numbers can be extrapolated to provide a figure for the total number of visits over the year, whereas in reality, many of the visitors recorded during the survey are likely to visit the Sizewell area only a few times during a year."	Argument made makes no sense. Many visitors might visit only a few times per year. However, on each day there will be a different tranche of occasional visitors. As the 500,000 is derived by extrapolating counts of people on a given day it is not precautionary.	
68	6.3 Volume 2 Main Development Site: Chapter 14 – Terrestrial Ecology and Ornithology	14.12.93	The main breeding areas for nightjar within the Minsmere-Walberswick SPA are over 1 km from the main development site, where the predicted peak noise levels from construction activities are considerably below the noise threshold where disturbance could occur.	Nightjar are likely to forage considerable distances from heaths and not sure impacts for foraging nightjar are addressed anywhere?	

Appendix 3:
Amphibian and Reptile Conservation (2020). Report

to
Written Submission

for the
Royal Society for the Protection of Birds
and Suffolk Wildlife Trust

Submitted for Deadline 2
2 June 2021

Planning Act 2008 (as amended)

In the matter of:

Application by NNB Generation Company (SZC) Limited for an Order Granting Development Consent for

The Sizewell C Project

Planning Inspectorate Ref: EN010012
RSPB Registration Identification Ref: 20026628
Suffolk Wildlife Trust Registration Identification Ref: 20026359

amphibian and reptile conservation

Sizewell C Project: comments on natterjack toad mitigation proposals

Report to Suffolk Wildlife Trust

Amphibian and Reptile Conservation

October 2020

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www.arc-trust.org

1. Introduction

This report contains comments from the Amphibian and Reptile Conservation Trust (hereafter, ARC) on the proposals for mitigating the impacts of the proposed Sizewell C new nuclear power station on natterjack toad *Epidalea calamita*. The report was commissioned by Suffolk Wildlife Trust (SWT), and written by Jim Foster (Conservation Director, ARC) with support from Yvette Martin (Amphibian Conservation Officer, ARC). The comments are based on the following information sources:

(a) Sizewell C proposal documents submitted by the developer (EDF) to the Planning Inspectorate, as available at the Sizewell C Development Consent Order (DCO) Document Portal, (sizewellcdco.co.uk). Hereafter these are referred to as the "EDF proposals". In particular, we have reviewed the following documents in depth (identified here by their filenames):

- SZC_Bk6_ES_V2_Ch14_Terrestrial Ecology Ornithology
- SZC_Bk6_ES_V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14A5_Amphibians
- SZC_Bk6_ES_V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species
- SZC_Bk6_ES_V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C7A_Natterjack_Toad_Mitigation_Strategy_Fig14C7A.1_14C7A.5
- SZC_Bk8_8.2_Outline_Landscape_and_Ecology_Management_Plan

(b) information on natterjack toads in Suffolk collected as part of the national monitoring programme co-ordinated by ARC.

The report also takes into account experience gained by the authors on natterjack toad conservation and management across their range in Great Britain.

2. Review of natterjack toad survey data and impact assessment

The natterjack toad survey data used as a basis for the EDF proposals are largely sufficient for this type of impact assessment and mitigation planning, and are generally well presented with the exception of surveys in terrestrial habitat. We agree with some of the conclusions about the origin and status of the natterjack toad population adjacent to the development footprint, i.e. the Retsom's Field population managed by SWT on behalf of EDF. The EDF proposals adequately summarise long-term monitoring data, and the suggestion of a population size of around 30 adults[1] is reasonable. However, it would have been appropriate to caveat this with a note about the fact that monitoring effort has not been consistent and intensive enough for a precise and reliable population estimate. We do not think it likely that the adults found in Pond N1 in 2006 and 2007 originated from Minsmere as suggested (1.2.13)[2], given that the Minsmere population has been in

[1] ES V2 Ch 14 Terrestrial Ecology Ornithology Appx 14A5 Amphibians, pdf pg 13, para 1.2.25 (doc pg: 9) (APP-233)
ES V2 Ch14 Terrestrial Ecology and Ornithology pg 140 para 14.10.5 (doc pg: 134) (AS-033)
ES V2 Ch14 Terrestrial Ecology Ornithology Appx14C Protected Species – pdf pg 330, para 1.2.15 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 9) (APP -252)

[2] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14A5_Amphibians – pdf pg 10, para 1.2.13 (doc pg:

a long-term decline and given the distance to that population. We do not think that the use of terrestrial habitat in Retsom's Field and surrounding areas has been well described in the EDF proposals.

Some of the statements in the EDF proposals about natterjack toads are unusual and call into question the author's level of expertise with this species. For instance, there are peculiar comments implying that natterjack toads might be considered poor swimmers (1.1.12)[3]; the intention of the comment about natterjack toads not being found during reptile surveys is unclear and requires further context (1.3.5)[4]. The mitigation proposals (see below for detailed comments) indicate at several points that conventions used in great crested newt mitigation have simply been applied to natterjack toads, despite the fact that there are marked differences in the ecology of these two species.

There are significant shortcomings with the impact assessment in the EDF proposals. The impacts stated in section 1.3 are all appropriate, but some possible impacts are not mentioned, and some are mentioned but are not adequately described.

The range of impact types (1.3) [5] is incomplete, as it apparently does not include consideration of the following:

- The likelihood that rabbit grazing would be reduced or possibly re-distributed by Water Management Zone (WMZ) construction activities such as noise and lighting, meaning that (a) ongoing maintenance of a low sward would be compromised, which is known to be a significant issue for natterjack toads; (b) the provision of bare ground and underground voids through rabbit warren excavations would be compromised, leading to a reduction in refuge availability.

- The possibility that WMZ construction and operation would result in significant changes to the hydrology of Retsom's Field and surrounding areas. If the field were to be either inundated, or conversely made significantly drier by the works, this would likely have an adverse effect on the natterjack toad population. We do not agree with the simplistic conclusion (1.3.8)[6] that hydrology is not an issue simply because the existing breeding pond (N1) is lined and the substrate is sandy. Were the Field to be inundated during the winter, there might be overwintering

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6) (APP-233)

[3] ES V2 Ch14 Terrestrial Ecology Ornithology Appx14C Protected Species – pdf pg 354, para 1.1.12 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 23) (APP-252)
ES_V2_Ch14_Terrestrial-Ecology-and-Ornithology – pdf pg 144, para 14.10.22 (doc pg: 138) (AS-033)

[4] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 358, para 1.3.5 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 27) (APP-252)

[5] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 341-342, para 1.3.1-1.3.10 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 10-11) (APP-252)

[6] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 342, para 1.3.8 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 11) (APP-252)

mortality, for example. If the field were to become significantly drier, this might impact the suitability of the substrate for natterjack toads, which require some water content in the substrate to avoid desiccation when taking refuge. Hydrological changes might affect prey availability. In addition, the possibility that hydrological changes might affect N4, which is unlined, has not been investigated.

The scale of the following impact is inadequately characterised:

- Loss of terrestrial habitat (1.3.6)[7]. We believe the impact of the "temporary" loss of terrestrial habitat during WMZ construction and operation has been significantly under-estimated and is not adequately explained. Natterjack toads are highly terrestrial, with most adults spending the majority of time on land and only a small fraction of the year in aquatic habitat; post-metamorphic immature individuals rarely if ever enter aquatic habitat. The exclusion of 3.5 ha of good quality terrestrial habitat close to the breeding pond for 3 generations could have a noticeable impact on the population through reduced foraging and refuge opportunities. The small size of the population, and its isolation, may exacerbate any negative effects.

Other comments on the impact assessment:

It is concerning that the wording regarding the location of WMZ boundary and to what extent it will actually impact on habitat in Retsom's Field is imprecise. There is mention of changes at the detailed design stage, but this makes it difficult to comment meaningfully on impacts now, since there is a degree of uncertainty.

There is a parcel of heathland to the far west of Retsom's Field, which would be lost to the WMZ. This area is likely to be used by natterjack toads, but the impact of its loss is not considered in the EDF proposals.

The particular vulnerabilities of this population, in conservation status terms, are not sufficiently addressed. This population is currently based on a single breeding pond, and if that were compromised the population would perish, resulting in a loss of range of significance at a much wider spatial scale. This needs to be considered in the context of the "FCS test" in the proposed mitigation licence.

We note that there is an error in the section describing the legislation applying to natterjack toads. At 1.1.2[8] the text indicates that that killing, injuring or taking

[7] ES V2 Ch14 Terrestrial Ecology Ornithology Appx14C Protected Species – pdf pg 341, para 1.3.6 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 10) (APP-252)

[8] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 350, para 1.1.2 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 19) (APP-252)

natterjack toads is an offence under the Wildlife and Countryside Act 1981 (as amended), but this is not the case. In fact, most provisions of Section 9 of that Act do not apply to natterjack toads, with the main protection being offered by The Conservation of Habitats and Species Regulations 2017 (as amended). We would expect factual matters like this to be correct in a mitigation report.

3. Review of natterjack toad mitigation and compensation proposals

The mitigation proposals appear largely to be based on methods and standard levels of effort used for great crested newts. Unfortunately they do not take into account some important ecological characters of natterjack toads, nor have they been sufficiently adjusted to account for this particular population. There is a disconnect between the section of the EDF proposals which discusses natterjack toad ecology (Appendix 14C.7A.3)[9], and the proposals for mitigation and compensation: the former do not appear to have sufficiently informed the latter. For example, it is unlikely that 30 days of capture effort would be sufficient to adequately deplete the population in the fenced area (Table 1.3)[10], and the apparent reliance on pitfall trapping is insufficient.

The post-development works are not adequately explained. For example, Table 1.4 is blank aside from row headings[11].

The proposed compensation for the loss of terrestrial habitats is inadequate. Presently the proposal is that a single new pond would be created, and that this would effectively generate new terrestrial habitat. However, the terrestrial habitat in the field is currently poor for natterjack toads, largely because the vegetation is rank. There is no meaningful mention of how the new breeding pond and terrestrial habitat would be maintained in suitable condition for natterjack toads after construction. Therefore there is a lack of confidence that sufficient terrestrial habitat would in fact be created and maintained.

It is notable that N3 has not yet been colonised; this is likely because of the small source population, distance to that source, and the poor surrounding terrestrial habitat. These issues would not be sufficiently addressed by the EDF proposals for the new pond, N5. However, it is in principle a sensible idea to create a “stepping stone” pond between N1 and N4.

In setting out any plans for new lined ponds, it has not been recognised that they have a finite lifespan, typically 20 years, though this can be much shorter depending on materials and local conditions. Pond liners may be punctured by

[9] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 353 App 14C.7A.3 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 22) (APP-252)
[10] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 346-347, Table 1.3 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 15-16) (APP-252)
[11] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 348, Table 1.4 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 17) (APP-252)

grazing animals and need repairing or replacing, for example. In addition, ponds need to be well designed, with shallow margins, and be maintained so that aquatic vegetation, competitors and predators do not proliferate. Measures have not been suggested as to how there will be continued provision of suitable breeding habitat.

The proposed linear feature (1.4.8, second bullet)[12] is unlikely to have any effect on connectivity. Natterjack toads are more likely to disperse over flat grassland, were
it to be of a suitable sward, than to use a linear mound for dispersal. This seems an odd statement to make. The feature would, however, create terrestrial refuge opportunities as stated, if well constructed.

The scale of the proposed compensation habitat is inadequate given the potential impacts. Given the significance of the population, its vulnerability in terms of size and isolation, and the risks posed by the development, it would be appropriate to create a larger extent of quality habitat, with the aim of extending the local distribution of the species and increasing its viability. That could buffer against any residual impacts caused by the construction and operational phase.

A cluster of ponds of slightly differing profiles would be more appropriate than a single pond. This would increase the chance that in any one year suitable breeding habitat would be available, buffering against changes in environmental conditions. The natterjack toad is a pioneer species and populations can decline rapidly if habitat (both terrestrial and aquatic) conditions are not adequately maintained. The EDF proposals do not give reassurance that there would be measures in place to ensure a sufficient extent and quality of habitat in the long term.

4. Review of natterjack toad monitoring proposals

The monitoring proposals are inadequate. It is not explained precisely what would be monitored, and the methods, effort and timing to achieve this. The lack of detail here makes it difficult to comment. For example, there is mention of monitoring being delivered annually during WMZ operation and biannually over a 6 year period afterwards[13], but in order to obtain a mark recapture estimate we would expect annual monitoring. Table 1.4 has blank spaces[14], it is unclear who has responsibility for monitoring, and there is no link to remedial work should monitoring identify an issue.

[12] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 343, para 1.4.8 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 13) (APP-252)
[13] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 345, para 1.4.12 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 14) (APP-252)
[14] ES-V2_Ch14_Terrestrial_Ecology_Ornithology_Appx14C_Protected_Species – pdf pg 348, Table 1.4 (doc pg: V2, C14, App 14C7A Natterjack Toad Mitigation Strategy pg 17) (APP-252)

5. Suggested changes to mitigation and compensation proposals

We suggest that the mitigation methods, effort and timing are revised to better take into account natterjack toad ecology, and the particular circumstances of this population. For example, during the capture phase, we would expect active capture by searching at night with a powerful torch, rather than a reliance on pitfall traps which appears to be the case with current proposals.

The compensation proposals should be prefaced by a clear ambition. We would like to see a commitment to securing a resilient population in the long term, based on the availability of several breeding ponds set in good quality terrestrial habitat, with a local distribution expansion to the north.

Habitat creation should involve a network of at least four new ponds, with slightly differing profiles and designed to provide good breeding habitat for natterjack toads. These should be sited in Retsom's Field, around the location currently proposed for N5, providing breeding opportunities between the existing breeding pond and N4.

When setting out habitat creation, management and monitoring proposals, the particular constraints of the area should be fully explored and explained. In particular, the soils and presence of waders may influence the type and location of new ponds.

There should be a comprehensive plan for the short-term and long-term management of both breeding ponds and terrestrial habitat, clearly showing objectives, routine management, remedial measures (e.g. pumping out of ponds), triggers for remedial measures, and an inspection regime. There should be clear responsibility for the actions listed, and a commitment for resourcing the actions.

6. Suggested changes to monitoring proposals

We would like to see significantly more detail on the monitoring proposals, in terms of objectives, methods, effort, timing, responsibility for delivery, use of monitoring results, and sharing of data.

In terms of monitoring objectives, we suggest that the following should be covered:

- Assessing the efficacy of mitigation measures put in place to reduce harm to natterjack toads during WMZ construction and operation
- Assessing the efficacy of habitat creation and enhancement measures in the short and long term.

In terms of field survey methods used, we suggest two broad categories are included:

> (1) standard monitoring methods for this species: counts of spawn strings laid, assessment of metamorph production, counts of adults in ponds during the breeding season, assessment of habitat quality;
>
> (2) a finer resolution of population status assessment: an estimate of adult population size through repeat surveys of adults and applying standard mark-recapture statistics, using natural markings to identify individuals (see, e.g. Smith et al 2018; McMurray undated).

Routine monitoring during the construction phase should be undertaken annually with at least 7 survey visits per year, as recommended for standard natterjack toad monitoring (e.g. see guidance on ARC's website at: https://www.arc-trust.org/targeted-surveys-for-natterjack-toads).

Monitoring should be undertaken annually for at least 6 years following completion of WMZ operation, in order to detect any residual impacts of the development and to trigger remedial actions. Monitoring should cover all the existing ponds (N1-4) and newly created ponds. There should be a clear statement of investigations and potential remedial actions should monitoring detect problems such as a failed pond liner; these investigations and remedial actions should be costed and secure resourcing plans put in place.

Assessment of habitat quality should include a note of management operations (e.g. grazing) and natural factors (e.g. rabbit activity, weather) likely to be influencing habitat condition.

Monitoring should be carried out by personnel meeting standard criteria (CIEEM, 2013).

Constraints on monitoring should be explored and explained. For example, the presence of waders may require restrictions on access.

Monitoring results should be shared with those with an interest or involvement in managing the natterjack toad population, and with the national monitoring programme operated by ARC.

7. Summary

EDF's proposals for mitigating and compensating the impacts of the proposed Sizewell C new nuclear power station on a population of natterjack toads *Epidalea calamita* are assessed. This species is strictly protected by law. The current status of the population is adequately described in EDF proposals. However, we find significant shortcomings in the proposals for mitigating and compensating impacts, and for monitoring the population. The framing of impacts as "temporary" when they would in fact last for three natterjack toad generations is a notable misinterpretation. There is insufficient reassurance that habitat creation and management proposals would generate and maintain a resilient population of natterjack toads and put in place suitable monitoring oversight. In general, the proposals do not adequately take into account the ecology and local conservation status of the population. Amendments to the proposals are suggested. In particular, we suggest that proposals need to be much clearer in relation to methods, effort, timing, responsibility for actions, remedial operations, and long-term resourcing.

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Appendix 4: References to Written Submission

for the Royal Society for the Protection of Birds and Suffolk Wildlife Trust

Submitted for Deadline 2
2 June 2021

Planning Act 2008 (as amended)

In the matter of:

Application by NNB Generation Company (SZC) Limited for an Order Granting Development Consent for

The Sizewell C Project

Planning Inspectorate Ref: EN010012
RSPB Registration Identification Ref: 20026628
Suffolk Wildlife Trust Registration Identification Ref: 20026359

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Citations/Conservation Objectives/Supplementary Advice/OLD

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Alde-Ore Estuary SPA Site Conservation Objectives

Alde-Ore Estuary SPA – Supplementary Advice on Conservation Objectives

Minsmere-Walberswick Ramsar Information Sheet

Minsmere-Walberswick SPA

Minsmere-Walbeswick SPA Site Conservation Objectives

Minsmere–Walberswick SPA - Supplementary Advice on Conservation Objectives

Minsmere to Walberswick Heaths and Marshes SAC

Minsmere to Walberswick Heaths and Marshes SAC Site Conservation Objectives

Minsmere-Walberswick Heaths and Marshes SSSI

Minsmere-Walberswick Heaths and Marshes SSSI Views About Management

Outer Thames Estuary SPA Natura 2000 Standard Data Form

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Sandlings SPA Site Conservation Objectives

Sizewell Marshes SSSI

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EC Directive 79/409 on the Conservation of Wild Birds: Special Protection Areas

Alde-Ore Estuary (Suffolk)

The Alde-Ore Estuary proposed Special Protection Area (pSPA) is situated on the east coast of Suffolk between Aldeburgh in the north and Bawdsey in the south. The site comprises the estuary complex of the rivers Alde, Butley and Ore, including Havergate Island and Orfordness. The variety of habitats important for breeding and wintering birds includes vegetated shingle, intertidal mudflats, semi-improved grazing marsh, saltmarsh and saline lagoons.

The site includes the entire Alde-Ore Estuary SSSI, notified in 1985 (revised in 1992 under the Wildlife and Countryside Act, 1981). The Alde-Ore Estuary SSSI includes the Orfordness-Havergate NNR, the English Nature owned part of which has already been designated as Orfordness-Havergate SPA.

The site qualifies under Article 4.1 of the EC Birds Directive by sustaining nationally important numbers of the following Annex 1 species, marsh harrier *Circus aeruginosus* (breeding), avocet *Recurvirostra avosetta* (wintering and breeding) ruff *Philomachus pugnax* (wintering), sandwich tern *Sterna sandvicensis* (breeding) and little tern *Sterna albifrons* (breeding). Further Annex 1 species winter on site, including, bittern *Botaurus stellaris*, Bewick's Swan *Cygnus columbianus*, hen harrier *Circus cyaneus*, golden plover *Pluvialis apricaria*, and short-eared owl *Asio flammeus*. Mediterranean gull *Larus melanocephalus*, common tern *Sterna hirundo* and Arctic tern *Sterna paradisaea* breed on Havergate Island.

The site qualifies under Article 4.2 of the Directive by regularly supporting internationally important numbers of two migratory species. The Orfordness colony of breeding lesser black-backed gull *Larus fuscus graellsii*, represented in 1995, 12% of the British population and 8% of the world population of the *graellsii* race. The five year wintering peak mean 1989/90 to 1993/94 for redshank *Tringa totanus*, was 1,662 birds, representing 1.5 % of the British population and 1.1% of the east Atlantic flyway population.

The site supports over 1% of the British wintering population of the following (calculated from five year winter peak means 1989/90 to 1993/94), shelduck *Tadorna tadorna*, wigeon *Anas penelope*, teal *Anas crecca*, black-tailed godwit *Limosa limosa*. In addition, the site supports over 1% of the British breeding population of, Gadwall *Anas strepera*, shoveler *Anas clypeata* and herring gull *Larus argentatus*.

The site also supports a notable assemblage of breeding and wintering wetland birds, in addition to the species mentioned above. Breeding species include, oystercatcher *Haematopus ostralegus*, ringed plover *Charadrius hiaticula*, lapwing *Vanellus vanellus* (also winter) black headed gull *Larus ridibundus* and barn owl *Tyto alba*. Wintering species include, cormorant *Phalacrocorax carbo*, European white-fronted goose *Anser abifrons albifrons*, brent goose *Branta bernicla*, pintail *Anas acuta*, grey plover *Pluvialis squatarola*, dunlin *Calidris alpina* and curlew *Numenius arquata*.

This citation / map relates to a site entered in the Register of European sites for Great Britain
Registe[redacted]
Date of
Signed
on behalf of the Secretary of State for the Environment

SPA Citation
January 1996

European Site Conservation Objectives for Alde–Ore Estuary Special Protection Area Site Code: UK9009112

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A081 *Circus aeruginosus*; Eurasian marsh harrier (Breeding)
A132 *Recurvirostra avosetta*; Pied avocet (Non-breeding)
A132 *Recurvirostra avosetta*; Pied avocet (Breeding)
A151 *Philomachus pugnax*; Ruff (Non-breeding)
A162 *Tringa totanus*; Common redshank (Non-breeding)
A183 *Larus fuscus*; Lesser black-backed gull (Breeding)
A191 *Sterna sandvicensis*; Sandwich tern (Breeding)
A195 *Sterna albifrons*; Little tern (Breeding)

This is a European Marine Site

This SPA is a part of the Alde Ore & Butley European Marine Site (EMS). These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

Alde-Ore Estuary SPA

Last updated: 13th September 2019

Supplementary advice

The Supplementary Advice on Conservation Objectives (SACOs) present attributes which are ecological characteristics or requirements of the classified species within a site. The listed attributes are considered to be those which best describe the site's ecological integrity and which if safeguarded will enable achievement of the Conservation Objectives.

Conservation Objectives relating to extent and distribution of habitat and population abundance are reflected in single attributes within the Supplementary Advice. Structure and function of habitats, and supporting processes for those habitats, are reflected in multiple attributes describing integrity of these ecological characteristics.

The Conservation Objective relating to the distribution of qualifying features (individual species or assemblages) may apply to most or all of the attributes listed in the SACOs and should be considered against them. Ensuring integrity of attributes relating to supporting habitats and processes should allow birds to distribute themselves optimally within (and, sometimes, outside) the SPA boundary. This is perhaps particularly relevant for food availability; extent and distribution of supporting habitat; quality of supporting habitat; predation; and disturbance caused by human activity.

Attributes have a target which is either quantified or qualified depending on the available evidence. The target identifies as far as possible the desired state to be achieved for the attribute. In many cases, the attribute targets show if the current objective is to either 'maintain' or 'restore' the attribute. The targets given for each attribute do not represent thresholds to assess the significance of any given impact in Habitats Regulation Assessments. You will need to assess this on a case-by-case basis using the most current information available.

Where there is no evidence to determine a marine feature's condition, a vulnerability assessment, which includes sensitivity and exposure information for features and activities in a site, has been used as a proxy for condition. Evidence used in preparing the SACO has been cited with hyperlinks included where possible. Where references have not been provided, Natural England has applied ecological knowledge and expert judgement.

Some, but not all, of these attributes can also be used for regular monitoring of the condition of the classified features. The attributes selected for monitoring the features, and the standards used to assess their condition, are listed in separate monitoring documents, which will be available from Natural England. As condition assessment information becomes available, the conservation advice package will be reviewed accordingly.

When to use

You should use this information, along with the conservation objectives and case-specific advice issued by Natural England when developing, proposing or assessing an activity, plan or project that may affect the site.

Any proposals or operations which may affect the site or its features should be designed so they do not adversely affect any of the attributes in the SACO or achievement of the conservation objectives.

Features:
Choose one or more features and/or their sub-features below by selecting the applicable boxes in the tree. This will show the relevant targets. Where a feature has sub-features this will be indicated with a greyed out triangle below, which can be expanded.

- ☐ Avocet (Recurvirostra avosetta), Breeding
- ☐ Avocet (Recurvirostra avosetta), Non-breeding
- ☐ Lesser black-backed gull (Larus fuscus), Breeding
- ☐ Little tern (Sternula albifrons), Breeding
- ☐ Marsh harrier (Circus aeruginosus), Breeding
- ☐ Redshank (Tringa totanus), Non-breeding
- ☐ Ruff (Calidris pugnax), Non-breeding
- ☐ Sandwich tern (Thalasseus sandvicensis), Breeding

Reset | Select all | Show attributes and targets for selected features

Attributes:
You can filter to show only targets for certain attributes by selecting one or more attributes from the list below (use ctrl click to select multiple). Note that only attributes for the features you have chosen are shown.

Feature target

'Maintain' targets do not preclude the need for management, now or in the future, to avoid a significant risk of damage or deterioration to the feature. The supporting and/or explanatory notes in the SACOs set out why the target was chosen and any relevant site based supporting information. This is based on the best available information, including that gathered during monitoring of the feature's current condition.

Please select a feature.

See further guidance on how to undertake an HRA for a plan or project on a European site (https://www.gov.uk/guidance/habitats-regulations-assessments-protecting-a-european-site).

These tables bring together the findings of the best available scientific evidence which may be updated or supplemented in further publications from Natural England and other sources. You may decide to use other additional sources of information.

These tables do not give advice about SSSI features or other legally protected species which may also be present within the European site.

Information Sheet on Ramsar Wetlands (RIS)

Categories approved by Recommendation 4.7 (1990), as amended by Resolution VIII.13 of the 8th Conference of the Contracting Parties (2002) and Resolutions IX.1 Annex B, IX.6, IX.21 and IX. 22 of the 9th Conference of the Contracting Parties (2005).

Notes for compilers:

1. The RIS should be completed in accordance with the attached *Explanatory Notes and Guidelines for completing the Information Sheet on Ramsar Wetlands*. Compilers are strongly advised to read this guidance before filling in the RIS.

2. Further information and guidance in support of Ramsar site designations are provided in the *Strategic Framework for the future development of the List of Wetlands of International Importance* (Ramsar Wise Use Handbook 7, 2nd edition, as amended by COP9 Resolution IX.1 Annex B). A 3rd edition of the Handbook, incorporating these amendments, is in preparation and will be available in 2006.

3. Once completed, the RIS (and accompanying map(s)) should be submitted to the Ramsar Secretariat. Compilers should provide an electronic (MS Word) copy of the RIS and, where possible, digital copies of all maps.

1. Name and address of the compiler of this form:

FOR OFFICE USE ONLY.

DD MM YY

Designation date | Site Reference Number

Joint Nature Conservation Committee
Monkstone House
City Road
Peterborough
Cambridgeshire PE1 1JY
UK
Telephone/Fax: +44 (0)1733 – 562 626 / +44 (0)1733 – 555 948
Email: RIS@JNCC.gov.uk

2. Date this sheet was completed/updated:

Designated: 05 January 1976

3. Country:

UK (England)

4. Name of the Ramsar site:

Minsmere–Walberswick

5. Designation of new Ramsar site or update of existing site:

This RIS is for: Updated information on an existing Ramsar site

6. For RIS updates only, changes to the site since its designation or earlier update:

a) Site boundary and area:

** Important note: If the boundary and/or area of the designated site is being restricted/reduced, the Contracting Party should have followed the procedures established by the Conference of the Parties in the Annex to COP9 Resolution IX.6 and provided a report in line with paragraph 28 of that Annex, prior to the submission of an updated RIS.

b) Describe briefly any major changes to the ecological character of the Ramsar site, including in the application of the Criteria, since the previous RIS for the site:

7. Map of site included:

Refer to Annex III of the *Explanatory Notes and Guidelines*, for detailed guidance on provision of suitable maps, including digital maps.

a) A map of the site, with clearly delineated boundaries, is included as:

i) **hard copy** (required for inclusion of site in the Ramsar List): *yes* ✓ -or- *no* □;
ii**) an electronic format** (e.g. a JPEG or ArcView image) Yes
iii) **a GIS file providing geo-referenced site boundary vectors and attribute tables** *yes* ✓ -or- *no* □;

b) Describe briefly the type of boundary delineation applied:

e.g. the boundary is the same as an existing protected area (nature reserve, national park etc.), or follows a catchment boundary, or follows a geopolitical boundary such as a local government jurisdiction, follows physical boundaries such as roads, follows the shoreline of a waterbody, etc.

The site boundary is the same as, or falls within, an existing protected area.

For precise boundary details, please refer to paper map provided at designation

8. Geographical coordinates (latitude/longitude):

52 18 55 N 01 38 02 E

9. General location:

Include in which part of the country and which large administrative region(s), and the location of the nearest large town.

Nearest town/city: Southwold

Composite site situated on the coast of Suffolk, between Southwold in the north and Sizewell in the south.

Administrative region: Suffolk

10. Elevation (average and/or max. & min.) (metres): 11. Area (hectares): 2018.92

Min. -1
Max. 24
Mean 9

12. General overview of the site:

Provide a short paragraph giving a summary description of the principal ecological characteristics and importance of the wetland.

This composite, Suffolk coastal site contains a complex mosaic of habitats, notably, areas of marsh with dykes, extensive reedbeds, mudflats, lagoons, shingle and driftline, woodland and areas of lowland heath. The site supports the largest continuous stand of reed in England and Wales and demonstrates the nationally rare transition in grazing marsh ditch plants from brackish to fresh water. The combination of habitats create an exceptional area of scientific interest supporting nationally scarce plants, British Red Data Book invertebrates and nationally important numbers of breeding and wintering birds.

13. Ramsar Criteria:

Circle or underline each Criterion applied to the designation of the Ramsar site. See Annex II of the *Explanatory Notes and Guidelines* for the Criteria and guidelines for their application (adopted by Resolution VII.11).

1, 2

14. Justification for the application of each Criterion listed in 13 above:

Provide justification for each Criterion in turn, clearly identifying to which Criterion the justification applies (see Annex II for guidance on acceptable forms of justification).

Ramsar criterion 1

The site contains a mosaic of marine, freshwater, marshland and associated habitats, complete with transition areas in between. Contains the largest continuous stand of reedbeds in England and Wales and rare transition in grazing marsh ditch plants from brackish to fresh water.

Ramsar criterion 2

This site supports nine nationally scarce plants and at least 26 red data book invertebrates.

Supports a population of the mollusc *Vertigo angustior* (Habitats Directive Annex II; British Red Data Book Endangered), recently discovered on the Blyth estuary river walls.

An important assemblage of rare breeding birds associated with marshland and reedbeds including: *Botaurus stellaris, Anas strepera, Anas crecca, Anas clypeata, Circus aeruginosus, Recurvirostra avosetta, Panurus biarmicus*

15. Biogeography (required when Criteria 1 and/or 3 and /or certain applications of Criterion 2 are applied to the designation):

Name the relevant biogeographic region that includes the Ramsar site, and identify the biogeographic regionalisation system that has been applied.

a) biogeographic region:

Atlantic

b) biogeographic regionalisation scheme (include reference citation):

Council Directive 92/43/EEC

16. Physical features of the site:

Describe, as appropriate, the geology, geomorphology; origins - natural or artificial; hydrology; soil type; water quality; water depth, water permanence; fluctuations in water level; tidal variations; downstream area; general climate, etc.

Soil & geology	acidic, neutral, shingle, sand, peat, nutrient-poor, mud, alluvium
Geomorphology and landscape	lowland, coastal, valley, floodplain, shingle bar, intertidal sediments (including sandflat/mudflat), open coast (including bay), estuary, lagoon
Nutrient status	mesotrophic
pH	circumneutral
Salinity	brackish / mixosaline, fresh, saline / euhaline
Soil	no information
Water permanence	usually permanent
Summary of main climatic features	Annual averages (Lowestoft, 1971–2000) (www.metoffice.com/climate/uk/averages/19712000/sites/lowestoft.html) Max. daily temperature: 13.0° C Min. daily temperature: 7.0° C Days of air frost: 27.8 Rainfall: 576.3 mm Hrs. of sunshine: 1535.5

General description of the Physical Features:

Minsmere – Walberswick comprises two large marshes, the tidal Blyth estuary and associated habitats. This composite coastal site contains a complex mosaic of habitats, notably areas of marsh with dykes, extensive reedbeds, mudflats, lagoons, shingle, woodland and areas of lowland heath. It supports the largest continuous stand of common reed *Phragmites australis* in England and Wales, and demonstrates the nationally rare transition in grazing marsh ditch plants from brackish to fresh water.

17. Physical features of the catchment area:

Describe the surface area, general geology and geomorphological features, general soil types, general land use, and climate (including climate type).

Minsmere – Walberswick comprises two large marshes, the tidal Blyth estuary and associated habitats. This composite coastal site contains a complex mosaic of habitats, notably areas of marsh with dykes, extensive reedbeds, mudflats, lagoons, shingle, woodland and areas of lowland heath.

18. Hydrological values:

Describe the functions and values of the wetland in groundwater recharge, flood control, sediment trapping, shoreline stabilization, etc.

No special values known

19. Wetland types:

Marine/coastal wetland

Code	Name	% Area
Other	Other	30
U	Peatlands (including peat bogs swamps, fens)	30
G	Tidal flats	12.9
E	Sand / shingle shores (including dune systems)	12.4
H	Salt marshes	7.2
M	Rivers / streams / creeks: permanent	4
F	Estuarine waters	2.5
J	Coastal brackish / saline lagoons	1

20. General ecological features:

Provide further description, as appropriate, of the main habitats, vegetation types, plant and animal communities present in the Ramsar site, and the ecosystem services of the site and the benefits derived from them.

This composite Suffolk coastal site contains a complex mosaic of habitats notably, areas of marsh with dykes, extensive reedbeds, mud flats, lagoons, shingle, woodland and areas of lowland heath. The site supports the largest continuous stand of reed *Phragmites australis* in England and Wales and nationally rare transition in grazing marsh ditch plants from brackish to fresh water. The combination of habitats create an exceptional area of scientific interest supporting nationally scarce plants, RDB invertebrates and nationally important numbers of breeding and wintering birds.

Ecosystem services

21. Noteworthy flora:

Provide additional information on particular species and why they are noteworthy (expanding as necessary on information provided in **12**. Justification for the application of the Criteria) indicating, e.g. which species/communities are unique, rare, endangered or biogeographically important, etc. *Do not include here taxonomic lists of species present – these may be supplied as supplementary information to the RIS.*

Nationally important species occurring on the site.

Higher Plants.

This is one of few sites nationally for red-tipped cudweed *Filago lutescens* (RDB2) which occurs on light, sandy soils.
The nationally rare species *Corynephorus canescens* (RDB3) occurs on coastal dune habitat.

The site supports a range of nationally scarce plant species characteristic of heathland, wetland and coastal habitats, and the transitions between them. *Althaea officinalis, Myriophyllum verticillatum, Ruppia cirrhosa, Sium latifolium, Sonchus palustris, Ceratophyllum submersum, Ranunculus baudotii,* and *Carex divisa* (all nationally scarce) are associated with reedbeds, grazing marsh or ditches. *Hordeum marinum* occurs on sea-walls, *Lathyrus japonicus* on coastal shingle, and *Crassula tillaea* on heathland.

22. Noteworthy fauna:

Provide additional information on particular species and why they are noteworthy (expanding as necessary on information provided in **12**. Justification for the application of the Criteria) indicating, e.g. which species/communities are unique, rare, endangered or biogeographically important, etc., including count data. *Do not include here taxonomic lists of species present – these may be supplied as supplementary information to the RIS.*

Birds

Species currently occurring at levels of national importance:

Species regularly supported during the breeding season:

Eurasian marsh harrier , *Circus aeruginosus*, Europe	16 pairs, representing an average of 10.5% of the GB population (5 year mean 1993-1997)
Mediterranean gull , *Larus melanocephalus*, Europe	2 apparently occupied nests, representing an average of 1.8% of the GB population (Seabird 2000 Census)
Black-headed gull , *Larus ridibundus*, N & C Europe	2558 apparently occupied nests, representing an average of 1.9% of the GB population (Seabird 2000 Census)
Little tern , *Sterna albifrons albifrons*, W Europe	20 apparently occupied nests, representing an average of 1% of the GB population (Seabird 2000 Census)

Species with peak counts in spring/autumn:

Great bittern , *Botaurus stellaris stellaris*, W Europe, NW Africa	3 individuals, representing an average of 3% of the GB population (5 year peak mean 1998/9-2002/3 - spring peak)
Eurasian teal , *Anas crecca*, NW Europe	3083 individuals, representing an average of 1.6% of the GB population (5 year peak mean 1998/9-2002/3)
Ruff , *Philomachus pugnax*, Europe/W Africa	10 individuals, representing an average of 1.4% of the GB population (5 year peak mean 1998/9-2002/3)
Black-tailed godwit , *Limosa limosa islandica*, Iceland/W Europe	846 individuals, representing an average of 5.4% of the GB population (5 year peak mean 1998/9-2002/3 - spring peak)
Spotted redshank , *Tringa erythropus*, Europe/W Africa	15 individuals, representing an average of 11% of the GB population (5 year peak mean 1998/9-2002/3)
Common greenshank , *Tringa nebularia*, Europe/W Africa	9 individuals, representing an average of 1.5% of the GB population (5 year peak mean 1998/9-2002/3)

Species with peak counts in winter:

Greater white-fronted goose , *Anser albifrons albifrons*, NW Europe	212 individuals, representing an average of 3.6% of the GB population (5 year peak mean for 1996/7-2000/01)

Gadwall , *Anas strepera strepera*, NW Europe	261 individuals, representing an average of 1.5% of the GB population (5 year peak mean 1998/9-2002/3)
Northern shoveler , *Anas clypeata*, NW & C Europe	238 individuals, representing an average of 1.6% of the GB population (5 year peak mean 1998/9-2002/3)
Hen harrier, *Circus cyaneus*, Europe	15 individuals, representing an average of 2% of the GB population (5 year peak mean 1985/6-1989/90)
Water rail , *Rallus aquaticus*, Europe	5 individuals, representing an average of 1.1% of the GB population (5 year peak mean 1998/9-2002/3)
Pied avocet , *Recurvirostra avosetta*, Europe/Northwest Africa	329 individuals, representing an average of 9.6% of the GB population (5 year peak mean 1998/9-2002/3)
European golden plover , *Pluvialis apricaria apricaria*, P. a. altifrons Iceland & Faroes/E Atlantic	4503 individuals, representing an average of 1.8% of the GB population (5 year peak mean 1998/9-2002/3)
Common redshank , *Tringa totanus totanus*,	1386 individuals, representing an average of 1.1% of the GB population (5 year peak mean 1998/9-2002/3)
Lesser black-backed gull , *Larus fuscus graellsii*,	905 individuals, representing an average of 1.4% of the GB population (5 year peak mean 1998/9-2002/3)

Species Information

Nationally important species occurring on the site.

Invertebrates.

Ethmia bipunctella, Aleochara inconspicua, Philonthus dimidiatipennis, Deltote bankiana, Cephalops perspicuus, Erioptera bivittata, E. meijerei, Gymnancycla canella, Pisidium pseudosphaerium, Archanara neurica, Heliothis viriplaca, Pelosia muscerda, Photedes brevilinea, Senta flammea, Herminea tarsicrinalis, Haematopota grandis, Tipula marginata, Podalonia affinis, Arctosa fulvolineata, Eucosma catroptana, E.maritima, Melissoblaptes zelleri, Pima boisduvaliella, Acrotophthalmus bicolor, Limonia danica, Telmaturus tumidulus, Vertigo angustior (a Habitats Directive Annex II species (S1014)).

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23. Social and cultural values:

Describe if the site has any general social and/or cultural values e.g. fisheries production, forestry, religious importance, archaeological sites, social relations with the wetland, etc. Distinguish between historical/archaeological/religious significance and current socio-economic values.

Aesthetic
Aquatic vegetation (e.g. reeds, willows, seaweed)
Environmental education/ interpretation
Livestock grazing
Non-consumptive recreation
Scientific research
Tourism

b) Is the site considered of international importance for holding, in addition to relevant ecological values, examples of significant cultural values, whether material or non-material, linked to its origin, conservation and/or ecological functioning? No

If Yes, describe this importance under one or more of the following categories:

i) sites which provide a model of wetland wise use, demonstrating the application of traditional knowledge and methods of management and use that maintain the ecological character of the wetland:

ii) sites which have exceptional cultural traditions or records of former civilizations that have influenced the ecological character of the wetland:

iii) sites where the ecological character of the wetland depends on the interaction with local communities or indigenous peoples:

iv) sites where relevant non-material values such as sacred sites are present and their existence is strongly linked with the maintenance of the ecological character of the wetland:

24. Land tenure/ownership:

Ownership category	On-site	Off-site
Non-governmental organisation (NGO)	+	+
Local authority, municipality etc.	+	
National/Crown Estate	+	
Private	+	+
Other	+	

25. Current land (including water) use:

Activity	On-site	Off-site
Nature conservation	+	+
Tourism	+	+
Recreation	+	+
Current scientific research	+	
Cutting of vegetation (small-scale/subsistence)	+	
Permanent arable agriculture		+
Grazing (unspecified)	+	
Flood control	+	
Transport route	+	+
Non-urbanised settlements	+	+

26. Factors (past, present or potential) adversely affecting the site's ecological character, including changes in land (including water) use and development projects:

Explanation of reporting category:

1. *Those factors that are still operating, but it is unclear if they are under control, as there is a lag in showing the management or regulatory regime to be successful.*
2. *Those factors that are not currently being managed, or where the regulatory regime appears to have been ineffective so far.*

NA = Not Applicable because no factors have been reported.

Adverse Factor Category	Reporting Category	Description of the problem (Newly reported Factors only)	On-Site	Off-Site	Major Impact?
Erosion	2	Coastal squeeze within the Blyth Estuary	+		+
Recreational/tourism disturbance (unspecified)	2	Trampling damage to vegetated shingle and driftline communities, and disturbance of little tern nesting habitat	+		+

For category 2 factors only.
What measures have been taken / are planned / regulatory processes invoked, to mitigate the effect of these factors?
Erosion - English Nature provides advice to the Environment Agency and coastal local authorities in relation to flood and coastal protection management. This will inform the development of the Suffolk Estuaries strategies and the second generation shoreline management plan.

Recreational/tourism disturbance (unspecified) - English Nature to work with owners/occupiers and regulatory authorities to develop a strategy to manage visitor pressure on Suffolk vegetated shingle. These measures are likely to include temporary fencing and provision of boardwalks as well as measures to increase visitor awareness about the sensitivity of the shingle habitat, for example by interpretation, wardening.

Is the site subject to adverse ecological change? YES

27. Conservation measures taken:

List national category and legal status of protected areas, including boundary relationships with the Ramsar site; management practices; whether an officially approved management plan exists and whether it is being implemented.

Conservation measure	On-site	Off-site
Site/ Area of Special Scientific Interest (SSSI/ASSI)	+	
National Nature Reserve (NNR)	+	
Special Protection Area (SPA)	+	
Land owned by a non-governmental organisation for nature conservation	+	
Management agreement	+	
Site management statement/plan implemented	+	

Area of Outstanding National Beauty (AONB)	+	+
Environmentally Sensitive Area (ESA)	+	+
Special Area of Conservation (SAC)	+	

b) Describe any other current management practices:
The management of Ramsar sites in the UK is determined by either a formal management plan or through other management planning processes, and is overseen by the relevant statutory conservation agency. Details of the precise management practises are given in these documents.

28. Conservation measures proposed but not yet implemented:

e.g. management plan in preparation; official proposal as a legally protected area, etc.

No information available

29. Current scientific research and facilities:

e.g. details of current research projects, including biodiversity monitoring; existence of a field research station, etc.

Fauna.
Numbers of migratory and wintering wildfowl and waders are monitored annually as part of the national Wetland Birds Survey (WeBS) organised by the British Trust for Ornithology, Wildfowl & Wetlands Trust, the Royal Society for the Protection of Birds and the Joint Nature Conservation Committee.

Flora.
NVC and vegetation monitoring, bird and invertebrate surveys/monitoring carried out on EN's NNRs, NT, SWT, RSPB reserves.

30. Current communications, education and public awareness (CEPA) activities related to or benefiting the site:

e.g. visitor centre, observation hides and nature trails, information booklets, facilities for school visits, etc.

Facilities at National Trust and Royal Society for the Protection of Birds reserves.

31. Current recreation and tourism:

State if the wetland is used for recreation/tourism; indicate type(s) and their frequency/intensity.

Activities, Facilities provided and Seasonality.
A popular area for tourists as it is an AONB and contains Minsmere bird reserve and Dunwich heath, both with toilets/shop/cafe. There are more visitors in the summer, however it well used throughout the year by walkers and bird watchers.

32. Jurisdiction:

Include territorial, e.g. state/region, and functional/sectoral, e.g. Dept. of Agriculture/Dept. of Environment, etc.

Head, Natura 2000 and Ramsar Team, Department for Environment, Food and Rural Affairs, European Wildlife Division, Zone 1/07, Temple Quay House, 2 The Square, Temple Quay, Bristol, BS1 6EB

33. Management authority:

Provide the name and address of the local office(s) of the agency(ies) or organisation(s) directly responsible for managing the wetland. Wherever possible provide also the title and/or name of the person or persons in this office with responsibility for the wetland.

Site Designations Manager, English Nature, Sites and Surveillance Team, Northminster House, Northminster Road, Peterborough, PE1 1UA, UK

34. Bibliographical references:

Scientific/technical references only. If biogeographic regionalisation scheme applied (see **15** above), list full reference citation for the scheme.

Site-relevant references

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Please return to: **Ramsar Secretariat, Rue Mauverney 28, CH-1196 Gland, Switzerland**
Telephone**: +41 22 999 0170** • Fax: **+41 22 999 0169** • email: **ramsar@ramsar.org**

EC Directive 79/409 on the Conservation of Wild Birds:
Special Protection Area

MINSMERE-WALBERSWICK (SUFFOLK)

The Minsmere-Walberswick proposed SPA contains areas of grazing marsh, extensive reedbeds, the estuary of the River Blyth, and areas of lowland heath and woodland. The boundaries of the site follows those of the Minsmere-Walberswick Heath and Marshes.SSSI.

Minsmere-Walberswick qualifies under Article 4.1, by supporting, in summer, nationally important breeding populations of the following Annex 1 species: 5 booming male bitterns *Botauris stellaris* (presumed to represent 5 breeding pairs; 22% of the British breeding population) ; 15 breeding female marsh harriers *Circus aeruginosus* (20% of British) ; 47 pairs of avocet *Recurvirostra avosetta* (12% of British) ; 32 pairs of little tern *Sterna albifrons* (1% of British): and 24 pairs of nightjar *Caprimulgus europaeus* (1% of British).

The site qualifies also under Article 4.1 by regularly supporting, in winter, a nationally important wintering population of hen harrier *Circus cyaneus* (15 individuals, 2% of the British wintering population).

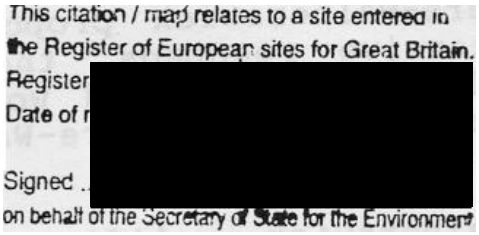
Minsmere-Walberswick qualifies under article 4.2 by supporting, in summer, in recent years, nationally important breeding populations of three regularly occurring migratory species: 24 pairs of gadwall *Anas strepera* (4% of British); 73 pairs of teal *A. crecca* (1% of British): and 23 pairs of shoveler *A. clvpeata* (2% of British) . Also notable is a nationally important breeding population of bearded tit *Panurus biarmicus* (50 pairs, 8% of British).

The site qualifies also under Article 4.2 by supporting nationally important wintering populations of three migratory waterfowl. (average peak counts for the five year period 1985/86 to 1989/90): 100 European white-fronted geese *Anser albifrons albifrons* (2% of the British wintering population); 90 gadwall *Anas strepera* (1% of British) , and 100 shoveler *Anas clypeata* (1% of British).

Minsmere-Walberswick is also of importance for an outstandingly diverse assemblage of breeding birds of marshland and reedbed habitats, including bittern, garganey *Anas querquedula*, marsh harrier, water rail *Rallus aquaticus*, Cetti's warbler *Cettia cetti* and Savi's warbler *Locustella lusciniodes*. Also notable is an assemblage of wintering waterfowl including, in addition to species listed above, Bewick's swan *Cygnus columbianus*, wigeon *Anas penelope*, teal *Anas crecca*, avocet; spotted redshank *Tringa erythropus*; and redshank *Tringa totanus*.

During severe winter weather Minsmere-Walberswick can assume even greater national and international importance as wildfowl and waders from many other areas arrive, attracted by relatively mild climate, compared with continental areas, and the abundant food resources available.

SPA Citation
HTR December 1991



European Site Conservation Objectives for Minsmere–Walberswick Special Protection Area Site Code: UK9009101

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A021 *Botaurus stellaris*; Great bittern (Breeding)
A051 *Anas strepera*; Gadwall (Non-breeding)
A051 *Anas strepera*; Gadwall (Breeding)
A052 *Anas crecca*; Eurasian teal (Breeding)
A056 *Anas clypeata*; Northern shoveler (Breeding)
A056 *Anas clypeata*; Northern shoveler (Non-breeding)
A081 *Circus aeruginosus*; Eurasian marsh harrier (Breeding)
A082 *Circus cyaneus*; Hen harrier (Non-breeding)
A132 *Recurvirostra avosetta*; Pied avocet (Breeding)
A195 *Sterna albifrons*; Little tern (Breeding)
A224 *Caprimulgus europaeus*; European nightjar (Breeding)
A394 *Anser albifrons albifrons*; Greater white-fronted goose (Non-breeding)

This is a European Marine Site

This SPA is a part of the Minsmere–Walberswick European Marine Site (EMS). These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

Minsmere-Walberswick SPA

Last updated: 13th September 2019

Supplementary advice

The Supplementary Advice on Conservation Objectives (SACOs) present attributes which are ecological characteristics or requirements of the classified species within a site. The listed attributes are considered to be those which best describe the site's ecological integrity and which if safeguarded will enable achievement of the Conservation Objectives.

Conservation Objectives relating to extent and distribution of habitat and population abundance are reflected in single attributes within the Supplementary Advice. Structure and function of habitats, and supporting processes for those habitats, are reflected in multiple attributes describing integrity of these ecological characteristics.

The Conservation Objective relating to the distribution of qualifying features (individual species or assemblages) may apply to most or all of the attributes listed in the SACOs and should be considered against them. Ensuring integrity of attributes relating to supporting habitats and processes should allow birds to distribute themselves optimally within (and, sometimes, outside) the SPA boundary. This is perhaps particularly relevant for food availability; extent and distribution of supporting habitat; quality of supporting habitat; predation; and disturbance caused by human activity.

Attributes have a target which is either quantified or qualified depending on the available evidence. The target identifies as far as possible the desired state to be achieved for the attribute. In many cases, the attribute targets show if the current objective is to either 'maintain' or 'restore' the attribute. The targets given for each attribute do not represent thresholds to assess the significance of any given impact in Habitats Regulation Assessments. You will need to assess this on a case-by-case basis using the most current information available.

Where there is no evidence to determine a marine feature's condition, a vulnerability assessment, which includes sensitivity and exposure information for features and activities in a site, has been used as a proxy for condition. Evidence used in preparing the SACO has been cited with hyperlinks included where possible. Where references have not been provided, Natural England has applied ecological knowledge and expert judgement.

Some, but not all, of these attributes can also be used for regular monitoring of the condition of the classified features. The attributes selected for monitoring the features, and the standards used to assess their condition, are listed in separate monitoring documents, which will be available from Natural England. As condition assessment information becomes available, the conservation advice package will be reviewed accordingly.

When to use

You should use this information, along with the conservation objectives and case-specific advice issued by Natural England when developing, proposing or assessing an activity, plan or project that may affect the site.

Any proposals or operations which may affect the site or its features should be designed so they do not adversely affect any of the attributes in the SACO or achievement of the conservation objectives.

Features:
Choose one or more features and/or their sub-features below by selecting the applicable boxes in the tree. This will show the relevant targets. Where a feature has sub-features this will be indicated with a greyed out triangle below, which can be expanded.

- ☐ Avocet (Recurvirostra avosetta), Breeding
- ☐ Bittern (Botaurus stellaris), Breeding
- ☐ Gadwall (Mareca strepera), Breeding
- ☐ Gadwall (Mareca strepera), Non-breeding
- ☐ Greater white-fronted goose (Anser albifrons albifrons), Non-breeding
- ☐ Hen harrier (Circus cyaneus), Non-breeding
- ☐ Little tern (Sternula albifrons), Breeding
- ☐ Marsh harrier (Circus aeruginosus), Breeding
- ☐ Nightjar (Caprimulgus europaeus), Breeding
- ☐ Shoveler (Spatula clypeata), Breeding
- ☐ Shoveler (Spatula clypeata), Non-breeding
- ☐ Teal (Anas crecca), Breeding

Reset | Select all | Show attributes and targets for selected features

Attributes:
You can filter to show only targets for certain attributes by selecting one or more attributes from the list below (use ctrl click to select multiple). Note that only attributes for the features you have chosen are shown.

Feature target

'Maintain' targets do not preclude the need for management, now or in the future, to avoid a significant risk of damage or deterioration to the feature. The supporting and/or explanatory notes in the SACOs set out why the target was chosen and any relevant site based supporting information. This is based on the best available information, including that gathered during monitoring of the feature's current condition.

Please select a feature.

See further guidance on how to undertake an HRA for a plan or project on a European site (https://www.gov.uk/guidance/habitats-regulations-assessments-protecting-a-european-site).

These tables bring together the findings of the best available scientific evidence which may be updated or supplemented in further publications from Natural England and other sources. You may decide to use other additional sources of information.

These tables do not give advice about SSSI features or other legally protected species which may also be present within the European site.

EC Directive 92/43 on the Conservation of Natural Habitats and of Wild Fauna and Flora

Citation for Special Area of Conservation (SAC)

Name:	Minsmere to Walberswick Heaths and Marshes
Unitary Authority/County:	Suffolk
SAC status:	Designated on 1 April 2005
Grid reference:	TM468682
SAC EU code:	UK0012809
Area (ha):	1265.52
Component SSSI:	Minsmere to Walberswick Heaths and Marshes SSSI

Site description:
Lowland dry heaths occupy an extensive area of this site on the east coast of England, which is at the extreme easterly range of heath development in the UK. The heathland is predominantly heather – western gorse (*Calluna vulgaris – Ulex gallii*) heath, usually more characteristic of western parts of the UK. This type is dominated by heather, western gorse and bell heather *Erica cinerea*.

Shingle beach forms the coastline at Walberswick and Minsmere. It supports a variety of scarce shingle plants including sea pea *Lathyrus japonicus*, sea campion *Silene maritima* and small populations of sea kale *Crambe maritima*, grey hair-grass *Corynephorus canescens* and yellow horned-poppy *Glaucium flavum*. A well-developed beach strandline of mixed sand and shingle supports annual vegetation. Species include those typical of sandy shores, such as sea sandwort *Honckenya peploides* and shingle plants such as sea beet *Beta vulgaris* ssp. *maritima*.

Qualifying habitats: The site is designated under **article 4(4)** of the Directive (92/43/EEC) as it hosts the following habitats listed in Annex I:

- Annual vegetation of drift lines
- European dry heaths
- Perennial vegetation of stony banks. (Coastal shingle vegetation outside the reach of waves)

This citation relates to a site entered in the Register of European Sites for Great Britain.
Register reference number: UK0012809
Date of registration: 14 June 2005
Signed: [redacted]
On behalf of the Secretary of State for Environment, Food and Rural Affairs

European Site Conservation Objectives for Minsmere to Walberswick Heaths and Marshes Special Area of Conservation Site Code: UK0012809

With regard to the SAC and the natural habitats and/or species for which the site has been designated (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the Favourable Conservation Status of its Qualifying Features, by maintaining or restoring;

- **The extent and distribution of qualifying natural habitats and habitats**
- **The structure and function (including typical species) of qualifying natural habitats, and**
- **The supporting processes on which qualifying natural habitats rely**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

H1210. Annual vegetation of drift lines

H1220. Perennial vegetation of stony banks; Coastal shingle vegetation outside the reach of waves

H4030. European dry heaths

This is a European Marine Site

This site is a part of the Minsmere–Walberswick European Marine Site. These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 as amended from time to time (the "Habitats Regulations"). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment', including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives and the accompanying Supplementary Advice (where available) will also provide a framework to inform the measures needed to conserve or restore the European Site and the prevention of deterioration or significant disturbance of its qualifying features.

These Conservation Objectives are set for each habitat or species of a Special Area of Conservation (SAC). Where the objectives are met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving Favourable Conservation Status for that species or habitat type at a UK level. The term 'favourable conservation status' is defined in regulation 3 of the Habitats Regulations.

Publication date: 27 November 2018 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

COUNTY: SUFFOLK SITE NAME: MINSMERE-WALBERSWICK HEATHS AND MARSHES

DISTRICT: SUFFOLK COASTAL/WAVENEY

Status: Site of Special Scientific Interest (SSSI) notified under Section 28 of the Wildlife and Countryside Act 1981, as amended

Local Planning Authority: SUFFOLK COASTAL DISTRICT COUNCIL, Waveney District Council, Suffolk County Council

National Grid Reference: TM 475645 Area: 2325.89 (ha.) 5747.27 (ac.)
TM 467772

Ordnance Survey Sheet 1:50,000: 156 1:10,000: TM 46 NE-NW-SW
TM 47 NE-NW-SE-SW

Date Notified (Under 1949 Act): See below Date of Last Revision: 1972

Date Notified (Under 1981 Act): 1989 Date of Last Revision: 1993

Other Information:
This site amalgamates Minsmere Level SSSI (notified in 1954), Walberswick SSSI (notified in 1954) and Brick Kiln Walks SSSI (notified in 1972).

Much of this site has been designated a Special Protection Area under EC Directive 79/409 on the Conservation of Wild Birds, and as a Wetland of International Importance under the Ramsar Convention.

Much of the site is included within 'A nature conservation review' by Ratcliffe (1977). It is within the Suffolk Coast and Heaths Area of Outstanding Natural Beauty.

Parts of the site are owned and/or managed as nature reserves and are listed below

Walberswick National Nature Reserve (English Nature)
Westleton Heath National Nature Reserve (English Nature)
Minsmere Reserve (Royal Society for the Protection of Birds)
Dunwich Heath (National Trust)
Norman Gwatkin Reserve (Suffolk Wildlife Trust)

Description and Reasons for Notification:
This composite site is situated on the coast of Suffolk between Southwold in the north and Sizewell in the south. It contains a complex series of habitats, notably mudflats, shingle beach, reedbeds, heathland and grazing marsh, which combine to create an area of exceptional scientific interest.

The tidal mudflats of the River Blyth estuary form sheltered feeding grounds for wildfowl and shorebirds, notably wigeon, shelduck, redshank and dunlin. Saltmarsh, dominated by sea purslane *Halimione portulacoides*, but also composed of sea

lavender *Limonium vulgare*, sea aster *Aster tripolium* and common cord-grass *Spartina anglica* fringes the southern shore of the estuary. Other saltmarsh species include glasswort *Salicornia* spp., sea rush *Juncus maritimus*, common saltmarsh grass *Puccinellia maritima* and sea couch-grass *Elymus pycnanthus*.

Shingle beach forms the coastline at Walberswick and Minsmere. This is subject to sea erosion and human disturbance but, nevertheless, it supports a variety of scarce shingle plants including sea pea *Lathyrus japonicus*, sea campion *Silene maritima* and small populations of sea kale *Crambe maritima*, grey hair-grass *Corynephorus canescens* and yellow horned-poppy *Glaucium flavum*. A narrow strip of yellow dune extends southwards at Minsmere behind which is a strip of dune grassland. A series of shallow, brackish lagoons and saltmarsh occurs behind the shingle beach between Walberswick and Dunwich.

Extensive reedbeds, consisting largely of pure stands of reed *Phragmites australis*, occur at Minsmere and Walberswick. These developed on former grazing marshes which were flooded as a war-time defence measure in 1940. Both marshes contain shallow pools of open water and are intersected by deep water channels. The reedbeds are an important habitat for birds and insects. There are large breeding populations of reed warbler and bearded tit. Other notable breeding species include marsh harrier, bittern, cetti's warbler, garganey and water rail. The marshes have a rich insect fauna; particularly moths, which includes a number of rare species: notably *Archanara neurica*, *Photedes brevilinea* and *Senta flammea*.

At Minsmere, a 20 hectare area of shallow lagoons and islands has been created for wading birds and wildfowl. This area is renowned for its breeding colony of avocets; shoveler, gadwall, teal and shelduck also breed.

Large blocks of grazing marsh are found near Eastbridge and Southwold. These marshes support a high number of species of breeding waterfowl such as snipe, redshank, gadwall, shoveler and black-tailed godwit. Dykes within the marshes contain very diverse aquatic plant communities, with brackish and freshwater types represented. Many nationally rare and scarce invertebrates such as the soldier fly *Odontomyia ornata* are found east of Eastbridge, as are a number of nationally scarce plants including sea barley *Hordeum marinum* and whorled water-milfoil *Myriophyllum verticillatum*. The marshes west of Eastbridge support a mosaic of different unimproved wetland communities including fen-meadow characterised by blunt-flowered rush *Juncus subnodulosus* and marsh thistle *Cirsium palustre*, reed beds, swamps dominated by lesser pond sedge *Carex acutiformis*, marshes dominated by meadowsweet *Filipendula ulmaria* with some angelica *Angelica sylvestris*, and alder *Alnus glutinosa* woodland.

High land at Minsmere, Westleton and Walberswick forms part of the East Suffolk Sandlings and is composed of infertile sands and gravels. This supports large areas of lowland heath, bracken, dry acidic grassland, woods and scrub.

Lowland heath, dominated by ling *Calluna vulgaris* but also containing bell heath *Erica cinerea* and cross-leaved heath *E. tetralix*, occupies a large continuous tract of about 400 ha at Minsmere, Dunwich and Westleton Heath with smaller areas at

Walberswick. This heathland provides a valuable habitat for two nationally decreasing birds, the. nightjar and woodlark.

Patches of unimproved acid grassland in which red fescue *Festuca rubra* and common bent *Agrostis capillaris* predominate, occur through the site but areas dominated by wavy hair-grass *Deschampsia flexuosa*, purple moor-grass *Molinia caerulea* and sand sedge *Carex arenaria* also occur. A variety of other acid grassland plants is also present, of which heath bedstraw *Galium saxatile* and sheep's sorrel *Rumex acetosella* are common. Scarce species include bird's-foot clover *Trifolium ornithopodioides* and mossy stonecrop *Crassula tillaea* together with a small colony of red-tipped cudweed *Filago lutescens*. There are also substantial areas dominated by bracken *Pteridium aquilinum* or gorse *Ulex europaeus* and *U. gallii*.

Mature plantation woodland, chiefly of oak *Quercus robur* or Scots pine *Pinus sylvestris* but also including sycamore *Acer pseudoplatanus* and sweet chestnut *Castanea sativa*, occur at Minsmere and Walberswick. Naturally regenerated woods of birch *Betula pendula* and Scots pine have arisen on former heathland and alder *Alnus glutinosa*, sallow *Salix* spp. and birch woodlands are also present on wet ground. This woodland and scrub provides important additional habitat diversity for birds and invertebrates.

Views About Management

ENGLISH NATURE

A statement of English Nature's views about the management of Minsmere-Walberswick Heaths & Marshes Site of Special Scientific Interest (SSSI).

This statement represents English Nature's views about the management of the SSSI for nature conservation. This statement sets out, in principle, our views on how the site's special conservation interest can be conserved and enhanced. English Nature has a duty to notify the owners and occupiers of the SSSI of its views about the management of the land.

Not all of the management principles will be equally appropriate to all parts of the SSSI. Also, there may be other management activities, additional to our current views, which can be beneficial to the conservation and enhancement of the features of interest.

The management views set out below do not constitute consent for any operation. English Nature's written consent is still required before carrying out any operation likely to damage the features of special interest (see your SSSI notification papers for a list of these operations). English Nature welcomes consultation with owners, occupiers and users of the SSSI to ensure that the management of this site conserves and enhances the features of interest, and to ensure that all necessary prior consents are obtained.

Management Principles

Blyth Estuary

Estuaries are relatively natural habitats which generally require little direct intervention to maintain them. Their proper management requires an understanding of the inputs and processes, both natural and otherwise, which affect them, and needs to be responsive to these factors.

The Blyth Estuary needs to be considered as a single functional unit that will respond to a variety of influences such as sea level rise, coastal squeeze and changing sediment regimes. The location and extent of habitats such as saltmarshes and mudflats depends upon the extent to which the estuary is constrained from responding to these influences. In the absence of constraints such as flood banks and hard defences, the estuary would adjust to sea level rise by inland translocation of intertidal habitats. Where constraints occur, space to accommodate greater volumes of water is compressed and the extent and quality of intertidal habitats declines.

The maintenance of high quality estuarine habitats such as saltmarshes and mudflats is essential if the associated plant and animal communities are to be maintained. In addition there are a variety of wetland habitats on the coastal flood plain that contribute to the overall importance of the estuary, including coastal grazing marshes,

brackish and freshwater pools and ditches. This combination of wetland and intertidal interest features, which may be dynamic over time, provides the feeding, roosting and nesting sites for the large numbers of breeding, wintering and passage wildfowl and waders.

Littoral sediments (mud and sand flats) of the Blyth Estuary

Intertidal mud and sand flats include a range of generally muddy or sandy low-gradient shores that are exposed to air during low tide and submerged during the higher tides. High energy shores, such as those on open coasts, are generally sandy in nature whilst more sheltered, low energy flats are muddier. They support a wide variety of marine invertebrates that represent an important food source for many fish and bird species.

Good water quality and sediment quality should be maintained, and the sediment budget within the estuarine or coastal system should not restricted by anthropogenic influences.

The birds that use mud and sandflats for feeding and roosting are vulnerable to disturbance from human activities, for example, bait digging, dog walking and wildfowling. These activities can lead to reduced time spent feeding, or individuals being restricted to areas with a poor food supply. Disturbance should therefore be minimised, especially at times when bird populations may be stressed, such as during severe winter weather.

The location and extent of mud or sandflats is dependent on the extent to which the estuary or coast where they occur is constrained from responding to sea level rise and changing sediment regimes. Management needs to create space to enable landward roll-back to take place in response to sea-level rise, and should also allow the system to be dynamic and retain the flexibility to respond to associated changes such as the movement of physical features within the system, e.g. migrating subtidal sandbanks.

Coastal saltmarsh

Saltmarshes form the upper vegetated portions of intertidal mudflats in sheltered coastal locations, such as estuaries, lagoons and beach plains. There is typically a zonation of vegetation, from plants adapted to regular immersion by the tides (halophytes), through to more widespread plant species in the areas less frequently covered by the sea. The halophyte plant species are confined to this type of habitat, and areas of structurally diverse vegetation provide good invertebrate habitat. Saltmarshes are also important nursery sites for several fish species, and important refuge, feeding and breeding grounds for wading birds and wildfowl.

Where saltmarshes require management this has traditionally been achieved by grazing, and previously used regimes should be continued. Grazing provides a variety of different habitats, particularly for wintering bird species, and if grazing were to cease there may be a loss of botanical diversity. The precise timing and intensity will vary according to local conditions and requirements, for example the type or availability of stock, or the need to avoid trampling ground nesting birds. However on many sites, the aim of will be to create a short turf that can be attractive to over-wintering wildfowl, with a reduction in stock density in the early summer for the

benefit of ground-nesting birds. Indeed, careful reduction of grazing can increase the number of breeding birds, without significantly altering the plant species composition. Care should be taken not to overgraze the site, as this may reduce the diversity of animal and plant species that the saltmarsh is able to support, as well as potentially impact the sediments supporting the saltmarsh.

Not all saltmarsh habitats require active management to retain their conservation interest. Where there has not been a history of grazing, the saltmarsh will be able to maintain itself and grazing-sensitive species are likely to be present, therefore grazing should not be introduced.

There are a number of factors that are contributing to saltmarsh change that management may need to take into consideration. These include coastal erosion as a result of coastal flood-defence works, rising sea-levels, variations in sediment deposition, and land claim for development.

Coastal lagoons

Coastal lagoons which occur behind the shingle beach between Dunwich and Walberswick are saline water bodies separated from the sea by a barrier (e.g. sand, shingle or rock sill), This separation from the sea makes them unique among coastal habitats and means that saline lagoons are either tideless, or where inlets occur, the tide has only a restricted effect on the lagoon. They retain part of their water-body at low tide, and this water may be either saline or brackish. They often support unusual assemblages of marine, estuarine and aquatic plants and animals, including lagoonal specialist species.

Any management needs to be carefully tailored to the needs of each individual lagoon and should be based on an understanding of the natural features of importance and the external factors affecting the lagoon. Indeed, where a lagoon is in a good and stable condition, active management is unlikely to be necessary. Maintaining salinity and water depths can be a key management priority, particularly where some lagoons become increasingly separated from the sea as a result of natural coastal processes - the balance between freshwater (e.g. from rainfall, streams or artificial outputs) and saline (i.e. sea water) inputs may change as a result. It may be necessary to actively manage freshwater and seawater input to favour certain species or communities. Whilst freshwater input is not essential to the conservation of lagoons, some connectivity with seawater is.

The water depth is also critical to many of the lagoonal specialist species with a depth between 0.5 and 1m being desirable. Some deeper water refuges are also beneficial. Siltation from surrounding land run-off may need to be addressed.

Water quality, and any direct and/or diffuse inputs from the surrounding land, can have a profound effect upon the productivity of lagoons and well-being of specialist species. Saline lagoons can show extreme reactions to a build up of some types of nutrients and therefore it may be necessary to actively manage inputs, especially where in close proximity to farmland.

In some cases, it may be desirable to allow vegetation to encroach into the lagoon to increase the diversity of habitats present, particularly for some breeding and migratory bird species. However vegetation should not be allowed to encroach to such an extent that it significantly reduces the areas of open water and shallow water, thus reducing the variety of habitats available to specialist species within the lagoon itself.

Islands in saline lagoons can be important for breeding birds and some management of the vegetation on these islands may be necessary to provide the best conditions for breeding birds. Two typical methods of vegetation control include flooding and hand clearance.

Vegetated shingle

Shingle is defined as sediment with particle sizes from 2-200mm. Shingle beaches form where sediment is first deposited on the shore by wave action. These deposits can then build up into more stable spits, bars or forelands. The types of vegetation that occur on shingle depend on the stability and structure of the shingle itself, but all must be able to cope with the unique physical and hydrological conditions typical of this habitat. This results in some communities being unique to shingle; including unusual moss- and lichen-rich communities that are of great conservation value. Shingle structures also provide important habitats for invertebrates and breeding birds.

A key management requirement is to avoid or minimise surface disturbance, especially in the more open communities. Many of the vegetation types and species associated with shingle are fragile and vulnerable to damage from trampling. This breaks up the fine humus that develops in the upper layers of the shingle that is vital for the plants to survive. Where recreational pressures are significant enough to result in the loss of vegetation cover, or prevent its recovery, it may be necessary to take steps to manage access. Disturbance of areas important for breeding birds should be minimised during the breeding season.

Where there is more closed vegetation cover, light grazing, by rabbits for example, may be all that is needed to prevent scrub encroachment on areas of grassland and heath. However, if there is a tradition of sheep grazing; it may be beneficial to continue this practice at a low intensity. In some cases grazing is not necessary, because of the low rates of plant growth on shingle structures, and can even be damaging, due to the fragility of shingle habitats. The introduction of grazing where it has not been traditionally practiced would not be beneficial.

Sand-dunes

Sand-dune habitats support a diverse range of species and communities. Sand-dunes develop where sand is blown landwards from the intertidal beach plain and is deposited above the high water mark. This is then colonised by dune building grasses which can continue to grow up as new layers of sand are deposited. A process of succession takes place as sand-dunes develop first into embryo dunes which can be ephemeral, then into semi-fixed dunes dominated by marram grass, and eventually into fixed dunes. Depending on the location of the dune in the system, different types of specialist vegetation occur. For example, the key features of the early-successional dunes are marram and lyme grass, with areas of bare sand. These more open communities can be important for amphibians and reptiles and some breeding birds

including several species of terns. The fixed dunes tend to be characterised by a more continuous sward of vegetation which may include dune grassland, low scrub, heath and lichen-dominated communities, which are often important for a variety of breeding and passage birds. Low-lying areas within dune systems are referred to as slacks, which can be either wet or dry and may be created by blowouts. These slacks are also important for amphibians, including rare species such as the natterjack toad.

The management of dune systems should take into account the need to maintain the range of habitats and associated species reflecting the different stages of succession, by maintaining (or restoring where necessary) the natural processes and dynamics of dune development and succession.

Dune management should aim to allow for all stages of the succession to be present on the site. Management of amenity beaches can affect the early stages of dune formation by removing the strandline that helps to trap blown sand and helps to develop new dune ridges. Dune systems exhibit a degree of dynamism, for example change from blowouts or newly deposited sand, which helps to retain a variety of successional stages within the site. Without management intervention, a mix of dune scrub and woodland may eventually replace the habitats on stable areas of the dune. Selective scrub management and grazing or mowing may be necessary. Where light grazing has traditionally been practised, this prevents the invasion of scrub and it should be continued. The effects of non-domestic grazing animals, such as rabbits, should also be taken into account. Other management options that might be appropriate include mowing to remove rank vegetation and in extreme situations can include turf stripping, which is useful for recreating the younger stages of slack formation.

Many of the vegetation types supported by sand dunes are fragile and vulnerable to erosion from heavy trampling. Where recreational pressures are significant enough to result in the loss of vegetation cover and prevent recovery, it may be necessary to take steps to manage access by putting boardwalks in or controlling activities in vulnerable areas such as the foredunes. It may also be necessary to manage access to limit the impacts of disturbance on breeding birds. Where recreation pressure is not severe, the impact of trampling can help to retain diversity on some sites – sandy tracks break up the vegetation sward and provide areas of bare sand thus increasing the diversity of habitats available.

Reedbed and Swamp

Reedbed and Swamp habitats develop on the fringes of open water, or in shallow depressions with permanent standing water. The plants may be rooted in the submerged soil or form a floating mat of inter-twined roots, rhizomes and stems. Swamps usually consist of a dominant single species of plant (e.g. reeds, tussock sedges, reedmace, reed sweet grass, reed canary grass and bull rushes) with a few other species thinly distributed among them. In common with most other types of wetland, swamps represent a transient stage in the change from open water to dry land.

Management should either seek to retain swamp communities in the same place or should acknowledge the dynamics of succession by ensuring there is always a new

niche for the swamp communities to develop in. The succession from swamp into floodplain fen, for example, as the diversity of species present increases, may be slowed by raising the water table and by periodically removing any encroaching scrub. If the vegetation surface of the whole wetland appears to be building up or drying out for some other reason it may be necessary to lower the ground level by creating scrapes or ponds. A programme of rotational cutting to maintain the reedbed may be necessary to encourage the vigorous growth of reed whilst preventing excessive build up of litter. Cutting should take place during the winter (November – March) and all cut material should be removed.

Management should ensure that appropriate water quality is maintained according to the requirements of the wetland communities present. Where swamp is in continuity with a waterbody, the water quality in the waterbody will affect the swamp. While some communities, such as reed swamp are unlikely to be very sensitive to nutritional enrichment, others, such as tussock sedge and narrow leaved reedmace, will be out-competed by other species (e.g. reed or reed sweet grass) where any increase in the amount of nutrients present occurs.

Swamp habitats have often survived where the vegetation has traditionally been cut for a variety of purposes, including use as building materials or animal bedding. It may be beneficial to consider re-instating these traditional management practices where they are not in conflict with other nature conservation objectives, such as the specific requirements of certain birds or invertebrates.

Wet grassland with breeding and wintering bird interest

Wet grasslands occur on land that is subject to periodic flooding or has a seasonally high water table and is waterlogged for much of the year. Wet grassland often supports a wide variety of plants and animals, in particular birds and invertebrates, and is an important habitat for breeding waders and wintering wildfowl.

Wet grassland requires active management if it is to retain its conservation interest. Generally, each year's growth of vegetation must be removed. Otherwise the sward becomes dominated by tall, vigorous grasses and rushes which, together with an associated build up of dead plant matter, suppress less vigorous species and lower the botanical richness of the sward. Traditionally, this management is achieved by grazing. Cattle are often the preferred stock, being relatively tolerant of wet conditions and able to control tall grasses and rank vegetation. Cattle also tend to produce a rather uneven, structurally diverse sward. However, ponies, or even hill sheep, can be used if necessary. Grazing usually takes place at times between late spring and early autumn, but the precise timing and intensity will depend on local conditions and requirements, such as the need to avoid trampling ground-nesting birds. Heavy poaching should be avoided but light trampling can be beneficial in breaking down leaf litter and providing areas for seed germination. Agricultural operations in general should be avoided before mid-June to minimise disturbance to breeding birds or the destruction of nests. An element of managed scrub, both within and fringing a field can be of importance to birds and invertebrates, as can a surrounding hedge.

Partial winter flooding is important in maintaining suitable habitat conditions for wintering birds. A mosaic of winter flooded grassland and permanently un-flooded grassland is desirable, with both temporary and permanent pools present. The maintenance of a mosaic of shallow surface pools and un-flooded areas during the winter will provide roosting and feeding habitat for wintering wildfowl and waders. From April onwards, the area of standing surface water should be reduced to increase the area available for nesting waders and also by concentrating aquatic invertebrates in small pools to provide suitable feeding areas for their young. Some shallow areas of flooding should be maintained until late June to provide patches of bare muddy ground on which the birds and their young can feed as raised sward height makes feeding on the drier areas more difficult. The birds using these features are directly vulnerable to disturbance, which can cause them to lose time spent feeding or drive them to areas with a poorer supply of food. Management should seek to minimise any harmful disturbance, especially at times when bird populations are under stress, such as severely cold conditions. Predators, especially crows and related species, should be controlled and this may be best achieved by limiting their nesting sites.

Careful maintenance of existing ditches and drains is usually acceptable practice, but abandonment or deepening of ditches can be harmful.

Ditches

Ditches are artificial habitats created by land drainage, or occasionally by the channelisation of small streams. They may represent the only remaining freshwater habitat within former wetland areas, and often support a wide range of aquatic plant and animal (in particular invertebrate) species that would have previously been more widespread in ponds and wetlands.

If left unmanaged, silt accumulates in the bottom of the ditches, and emergent plants such as reeds are able to colonise across the width of the ditch, leading to a loss of aquatic plant diversity and a gradual drying out of the ditch. To prevent this, periodical removal of sediment and vegetation may be necessary to return the ditch to an early stage of the management cycle. Ideally, ditch management should be undertaken on a rotation, creating a series of different management stages across a site at any one time. All stages of the management cycle have wildlife interest; recently cleared ditches are good for plants and animals which favour newly created habitats and cannot tolerate competition with other species; middle-stage ditches support a rich aquatic plant flora; and late-stage ditches may be important for a variety of invertebrates. The removal of both sediment and vegetation is usually better than simply cutting the vegetation, which does not recreate the earliest stages of the ditch management cycle. Where possible, management should aim to create shallow shelving margins rather than steep ditch sides. Where water voles are known to be present, the relevant good practice guidelines for ditch management and conservation should be followed.

Most ditch systems are subject to water level control, which should be managed to ensure that there is a sufficient depth of water (0.3-0.5m) in ditches throughout the year. Rapid or extreme changes in water level should be avoided unless they are known to be important to plant or animal communities relying on such fluctuations.

Ditches are susceptible to increased levels of nutrients which can cause a loss of aquatic plants and increases in algal growth. Other activities that can lead to this include the control or removal of aquatic plants, or the introduction of species such as bottom feeding coarse fish which uproot plants and disturb ditch sediments. Ditches are also susceptible to invasion by non-native aquatic plants such as floating pennywort and water fern, which are able to grow rapidly taking up available habitat and smothering other plants. Some native plants including a number of duckweed species are also able to take over in this way (although such growths are usually exacerbated by increased nutrients in the water) and management may be necessary to control such invasions where they cause a problem.

An important feature of many coastal ditch systems relating to water quality is the transition between freshwater and brackish water conditions, which often results in interesting changes in plant and animal species along ditch lengths or across the site. Management should aim to maintain such freshwater-brackish transitions where they occur.

Dry and wet lowland heath

On this site both wet and dry heath communities occur, however wet heath is restricted to a few isolated fragments, such as at Walberswick Common.. Heathland supports the greatest diversity of plants and animals (including a diverse invertebrate fauna and a number of characteristic bird species) where management maintains the open nature of the heath and by promoting a varied structure of uneven-aged stands of native heathers and other characteristic plants. It is generally beneficial if all stages of the heather life cycle are present. Without such management, heathland becomes progressively dominated by bracken, gorse and, on wet ground, purple moor grass tussocks. Eventually scrub and trees will invade. The precise management requirements will vary both between and within sites according to the needs of the different heathland interests present and site conditions.

Low intensity grazing is a suitable means of managing areas of dry heath. Generally areas of wet heath require limited management but light grazing may also be useful for maintaining the variation in vegetation composition and structure, and for controlling invasive grasses such as purple-moor grass. By feeding selectively in different areas and on different plants, free-roaming livestock help to maintain variation in the vegetation composition and structure. They can also suppress scrub encroachment and provide some light poaching to create small pockets of bare peat and sandy ground that are of benefit to a variety of specialised plants, invertebrates and reptiles. Sheep grazing is an acceptable method of management but cattle or hardy ponies may also be used, although care must be taken to avoid damage to the heather by trampling. An appropriate stocking rate should take into account local conditions and the timing and length of grazing, but an off-take of between 30-40% of the current growth increment is desirable. Heavy grazing should be avoided on wet heath as it can lead to a decline in characteristic dwarf shrub cover in favour of grass and sedge species, as well as excessive poaching and erosion of the underlying peat.

Alternatively, cutting or mowing may be useful options for managing dry heath where a mosaic of patches of heather of different ages is desired. The cut material should be removed to avoid nutrient accumulation on site and to allow the cut plants to re-sprout

successfully. However, mowing or cutting may not be suitable on wet heath or on mature stands of dry heath of importance for rare reptiles.

The rare silver-studded blue butterfly present on this site requires the maintenance of a very short heather turf and warm ground conditions associated with pioneer heath communities. Grazing is usually a suitable means of managing dry heath and can extend the time that the heath is in the pioneer phase. However, it is not usually capable on its own of maintaining the open conditions required by the silver-studded blue. These conditions are best achieved by either regular mowing to prevent the build up of mature vegetation, or by cutting patches or strips of vegetation on rotation. The cut material should be removed to avoid nutrient accumulation on site and to allow the cut plants to re-sprout successfully. Heathland supporting this rare butterfly should not be burnt.

Prescribed burning can be a useful tool for maintaining the structural diversity of some dry heathlands and for re-establishing areas of pioneer heath required by certain species, but special care is required when sensitive species are present (see above) and burning should not be used on wet heath vegetation. Burning must be used with caution, as inappropriate burning can be very damaging to both plant and animal communities and careful consideration should be given to the timing of the burn.

There is some benefit in retaining a few scattered individual trees and some small patches of scrub. For example, the maintenance of scattered mature Scots pine in undisturbed locations will provide suitable nest sites for hobbies. However, this should not encroach on the open nature of the habitat, and mechanical control or manual cutting followed by the careful application (spot application on areas of wet heath) of a suitable herbicide may be necessary to prevent this. Bracken invasions may need to be controlled in the same way.

Where gorse is present, scattered stands with a bushy structure rather than large continuous blocks are of greater benefit to the characteristic bird and invertebrate species associated with gorse scrub. For example, Dartford warbler require areas of open heath (with less than 25 trees per hectare) with over 50% cover of mature heather (preferably over 30 cm tall) and patches of dense, compact, mature gorse bushes (0.5-3 m tall) to be maintained. Winter cutting of ‘leggy’ stands of gorse and the removal of cut material will maintain gorse at different stages of re-growth and avoid nutrient accumulation in the soil.

Although careful maintenance of existing ditches and drains is usually acceptable, the abandonment or deepening of ditches or drains should be avoided. Water levels within areas of wet heath should be maintained to avoid adverse changes to the characteristic plant composition of the habitat. In some instances it may be appropriate to restore natural drainage where this is possible.]

Lowland acid grassland

Free-draining, acidic soil is the key requirement of the grassland communities at this site, but their maintenance also depends on active management. If neglected, the sward becomes dominated by tall, vigorous grasses or bracken which, together with an associated build up of dead plant matter, suppress less vigorous species and reduce

the botanical richness of the site. Eventually the sward reverts to scrub and even woodland. Traditionally, management has consisted of stock grazing and this remains the most appropriate management tool. Grazing, through the removal of plant matter and nutrients, helps to maintain an open sward of small tussocky grasses. It also, through disturbance and trampling, creates areas of open ground suitable for colonization by the lichens, ephemeral plants and invertebrates that are often characteristic of this type of grassland. However, rabbit grazing, though difficult to control, can also be a useful management tool in some situations. Occasional management of invasive scrub and bracken may be necessary.

Lowland wet woodland
Wet woodland includes a range of different woodland types but usually is dominated by ash, alder and willow species. It often supports important invertebrate species and assemblages.

Areas usually benefit from minimum intervention and are often best left undisturbed to limit damage to their fragile soils. This allows the development of old stands where individual trees reach maturity and die naturally to create gaps in the canopy, leading to a diverse woodland structure. However, works to remove dangerous trees in areas of public access may be necessary.

Where particularly important light-demanding or glade species interests are present, including where the woodland is spreading on to valuable open wetland habitat, it may be necessary to periodically clear areas of vegetation. In some woods a more active programme of management by coppice may be appropriate, where this has been the historical management and the conditions are such that it will not lead to heavy ground disturbance.

Broadleaved semi-natural woodland
There are many different ways in which broadleaved woodland can be managed to conserve its value for wildlife. The following gives broad views on a range of regimes that may be appropriate on your site.

A diverse woodland structure, with open space, a dense understory, and a more mature overstory is important. A range of ages and species within and between stands is desirable. Some dead and decaying wood, such as fallen logs, can provide habitats for fungi and invertebrates. However, work may be needed to make safe dangerous trees in areas of high public access. Both temporary and permanent open spaces benefit groups of invertebrates such as butterflies. They may require cutting to keep them open, and should be of sufficient size to ensure that sunny conditions prevail for most of the day.

Felling, thinning or coppicing may be used to create or maintain variations in the structure of the wood, and non-native trees and shrubs can be removed at this time. To avoid disturbance to breeding birds the work is normally best done between the beginning of August and the end of February. Work should be avoided when the ground is soft, to prevent disturbing the soil and ground flora. Normally successive felling, thinning or coppicing operations should be spread through the wood to promote diversity, but where there is open space adjacent plots should be worked to

encourage the spread of species that are only weakly mobile. Natural regeneration from seed or stump regrowth is preferred to planting because it helps maintain the local patterns of species and the inherent genetic character of the site.

Deer management and protection from rabbits or livestock are often necessary. Whilst light or intermittent grazing may increase woodland diversity, heavy browsing can damage the ground flora and prevent successful regeneration. Invasive species, such as *Rhododendron* or Himalayan balsam, should be controlled.

Parts of a wood should be left unmanaged to benefit species that do best under low disturbance or in response to natural processes. Within these areas some trees will eventually die naturally and dead wood accumulate.

All habitats

The habitats within this site are highly sensitive to inorganic fertilisers and pesticides, applications of which should be avoided both within the site itself and in adjacent surrounding areas. Herbicides may be useful in targeting certain invasive species, but should be used with extreme care. Access to this site, and any recreational activities within, may also need to be controlled.

NATURA 2000 – STANDARD DATA FORM

Special Protection Areas (SPAs) classified under Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds (codified version), also known as the 'Birds Directive'

and

Special Areas of Conservation (SACs) (includes candidate SACs, Sites of Community Importance (SCIs) and designated SACs) designated under Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora, also known as the 'Habitats Directive'

Each Natura 2000 site in the United Kingdom has its own Standard Data Form containing site-specific information.

The information provided here follows the officially agreed site information format for Natura 2000 sites, as set out in the Official Journal of the European Union recording the Commission Implementing Decision of 11 July 2011 (2011/484/EU).

The Standard Data Forms are generated automatically for all of the UK's Natura 2000 sites using the European Environment Agency's Natura 2000 software. The structure and format of these forms is exactly as produced by the EEA's Natura 2000 software (except for the addition of this coversheet and the end notes). The content matches exactly the data submitted to the European Commission.

Please note that these forms contain a number of codes, all of which are explained either within the data forms themselves or in the end notes.

Further technical documentation may be found here:
http://bd.eionet.europa.eu/activities/Natura_2000/reference_portal

In December 2015, several sections of the UK's previously published Standard Data Forms were updated. For details of the approach taken by the UK in this submission please refer to the following document:
http://jncc.defra.gov.uk/pdf/Natura2000_StandardDataForm_UKApproach_Dec2015.pdf.
These changes formed part of the UK Submission to the European Commission on 22/12/2015.

More general information on Special Protection Areas (SPAs) and Special Areas of Conservation (SACs) in the United Kingdom, including in Gibraltar, is available from the SPA homepage and SAC homepage on the JNCC website. These webpages also provide links to Standard Data Forms for all Natura 2000 sites in the UK.

Date Standard Data Form generated by the Joint Nature Conservation Committee:	14th November 2017 (UK Tranche 56)

NATURA 2000 - STANDARD DATA FORM

For Special Protection Areas (SPA),
Proposed Sites for Community Importance (pSCI),
Sites of Community Importance (SCI) and
for Special Areas of Conservation (SAC)

SITE **UK9020309**

SITENAME **Outer Thames Estuary**

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1. SITE IDENTIFICATION

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1.1 Type	1.2 Site code
A	UK9020309

1.3 Site name

Outer Thames Estuary

1.4 First Compilation date	1.5 Update date
2010-08	2017-11

1.6 Respondent:

Name/Organisation:	Joint Nature Conservation Committee
Address:	Joint Nature Conservation Committee Monkstone House City Road Peterborough PE1 1JY
Email:	

1.7 Site indication and designation / classification dates

Date site classified as SPA:	2010-08
National legal reference of SPA designation	Regulations 12A and 13-15 of the Conservation of Habitats and Species Regulations 2010 (http://www.legislation.gov.uk/uksi/2010/490/contents/made) as amended by The Conservation of Habitats and Species (Amendment) Regulations 2011 (http://www.legislation.gov.uk/uksi/2011/625/contents/made), and Regulations 12, 16 and 17 of the Offshore Marine Conservation (Natural Habitats, &c.) Regulations 2007 (http://www.legislation.gov.uk/uksi/2007/1842/contents/made).

2. SITE LOCATION

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2.1 Site-centre location [decimal degrees]:

Longitude	Latitude
1.545	51.916

2.2 Area [ha]:

392451.66

2.3 Marine area [%]

100.0

2.4 Sitelength [km]:

0.0

2.5 Administrative region code and name

NUTS level 2 code	Region Name
UKZZ	Extra-Regio

2.6 Biogeographical Region(s)

Atlantic (100.0 %)

3. ECOLOGICAL INFORMATION

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3.2 Species referred to in Article 4 of Directive 2009/147/EC and listed in Annex II of Directive 92/43/EEC and site evaluation for them

Species					Population in the site						Site assessment			
G	Code	Scientific Name	S	NP	T	Size		Unit	Cat.	D.qual.	A|B|C|D	A|B|C		
						Min	Max				Pop.	Con.	Iso.	Glo.
B	A001	Gavia stellata			w	6466	6466	i		G	A		C	
B	A195	Sterna albifrons			r	746	746	i		G	A		C	
B	A193	Sterna hirundo			r	532	532	i		G	B		C	

- **Group:** A = Amphibians, B = Birds, F = Fish, I = Invertebrates, M = Mammals, P = Plants, R = Reptiles
- **S:** in case that the data on species are sensitive and therefore have to be blocked for any public access enter: yes
- **NP:** in case that a species is no longer present in the site enter: x (optional)
- **Type:** p = permanent, r = reproducing, c = concentration, w = wintering (for plant and non-migratory species use permanent)
- **Unit:** i = individuals, p = pairs or other units according to the Standard list of population units and codes in accordance with Article 12 and 17 reporting (see reference portal)
- **Abundance categories (Cat.):** C = common, R = rare, V = very rare, P = present - to fill if data are deficient (DD) or in addition to population size information

- **Data quality:** G = 'Good' (e.g. based on surveys); M = 'Moderate' (e.g. based on partial data with some extrapolation); P = 'Poor' (e.g. rough estimation); VP = 'Very poor' (use this category only, if not even a rough estimation of the population size can be made, in this case the fields for population size can remain empty, but the field "Abundance categories" has to be filled in)

4. SITE DESCRIPTION

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4.1 General site character

Habitat class	% Cover
N02	1.5
N01	98.45
N03	0.05
Total Habitat Cover	100

Other Site Characteristics

3 Marine: Geology: mud,sand,gravel 4 Marine: Geomorphology: range of mobile sediments,tidal current stream

4.2 Quality and importance

ARTICLE 4.1 QUALIFICATION (79/409/EEC) Over winter the area regularly supports: Gavia stellata (North-western Europe - wintering) - 38% of the population in Great Britain peak mean over the period 1989-2006/07 The area supports breeding populations of: Sternula albifrons (in breeding season) - 19.64% of GB population (2011 - 2015) Sterna hirundo (in breeding season) - 2.66% of GB population (2011 - 2015)

4.3 Threats, pressures and activities with impacts on the site

The most important impacts and activities with high effect on the site

Negative Impacts			
Rank	Threats and pressures [code]	Pollution (optional) [code]	inside/outside [i|o|b]
H	C03		B
L	F02		I
L	H03		B
M	D03		B
L	G04		b

Positive Impacts			
Rank	Activities, management [code]	Pollution (optional) [code]	inside/outside [i|o|b]

Rank: H = high, M = medium, L = low
Pollution: N = Nitrogen input, P = Phosphor/Phosphate input, A = Acid input/acidification,
T = toxic inorganic chemicals, O = toxic organic chemicals, X = Mixed pollutions
i = inside, o = outside, b = both

4.5 Documentation

JNCC's weblink 'http://jncc.defra.gov.uk/page-1414' provides general information on marine SPAs. The weblink 'http://jncc.defra.gov.uk/page-6895' allows access to site specific information for all marine MPAs in UK offshore waters. See the UK Approach document for more information (link via the JNCC website).

Link(s): http://jncc.defra.gov.uk/pdf/Natura2000_StandardDataForm_UKApproach_Dec2015.pdf

http://jncc.defra.gov.uk/page-1414
http://jncc.defra.gov.uk/page-6895

5. SITE PROTECTION STATUS (optional)

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5.1 Designation types at national and regional level:

Code	Cover [%]	Code	Cover [%]	Code	Cover [%]
UK00	93.0	UK04	3.0	UK05	4.0

6. SITE MANAGEMENT

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6.1 Body(ies) responsible for the site management:

Organisation:	Southwold Harbour Authority – Waveney District Council, Ipswich Port Authority, Felixstowe Dock & Railway Company, Harwich Haven Authority, Brightlingsea Harbour Commissioners, Maldon Harbour Improvement Commissioners
Address:	
Email:	

Organisation:	For information about relevant management offshore please contact JNCC
Address:	
Email:	

Organisation:	Crouch Harbour Authority, Port of London Authority, Peel Ports London Medway, Great Yarmouth Local/District Unitary Authority, Waveney Local/District Unitary Authority, Suffolk Coastal Local/District Unitary Authority
Address:	
Email:	

Organisation:	Thanet Local/District Unitary Authority
Address:	
Email:	

Organisation:	Maldon Local/District Unitary Authority, Rochford Local/District Unitary Authority, Southend on Sea Local/District Unitary Authority, Swale Local/District Unitary Authority, Canterbury Local/District Unitary Authority
Address:	
Email:	

Organisation:	Natural England, Marine Management Organisation, Kent and Essex Inshore Fisheries & Conservation Authority, Crown Estate, Great Yarmouth Port Authority, Lowestoft – Associated British Ports (ABP)
Address:	
Email:	

6.2 Management Plan(s):

An actual management plan does exist:

- [ ] Yes
- [ ] No, but in preparation
- [X] No

6.3 Conservation measures (optional)

For available information on relevant conservation measures of the site, including the Conservation Objectives, see section 4.5.

7. MAP OF THE SITES

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INSPIRE ID:

Map delivered as PDF in electronic format (optional)

☐ Yes ☒ No

Reference(s) to the original map used for the digitalisation of the electronic boundaries (optional).

EXPLANATION OF CODES USED IN THE NATURA 2000 STANDARD DATA FORMS

The codes in the table below are also explained in the official European Union guidelines for the Standard Data Form. The relevant corresponding page number is shown in the table below.

1.1 Site type

CODE	DESCRIPTION	PAGE NO
A	SPA (classified Special Protection Area)	53
B	cSAC, SCI or SAC (candidate Special Area of Conservation, Site of Community Importance, designated Special Area of Conservation)	53
C	SPA area/boundary is the same as the cSAC/SCI/SAC i.e. a co-classified/designated site (Note: in the UK Natura 2000 submission, this is only used in Gibraltar)	53

3.1 Habitat representativity

CODE	DESCRIPTION	PAGE NO
A	Excellent representativity	57
B	Good representativity	57
C	Significant representativity	57
D	Non-significant presence	57

3.1 Habitat code

CODE	DESCRIPTION	PAGE NO
1110	Sandbanks which are slightly covered by sea water all the time	57
1130	Estuaries	57
1140	Mudflats and sandflats not covered by seawater at low tide	57
1150	Coastal lagoons	57
1160	Large shallow inlets and bays	57
1170	Reefs	57
1180	Submarine structures made by leaking gases	57
1210	Annual vegetation of drift lines	57
1220	Perennial vegetation of stony banks	57
1230	Vegetated sea cliffs of the Atlantic and Baltic Coasts	57
1310	Salicornia and other annuals colonizing mud and sand	57
1320	Spartina swards (Spartinion maritimae)	57
1330	Atlantic salt meadows (Glauco-Puccinellietalia maritimae)	57
1340	Inland salt meadows	57
1420	Mediterranean and thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi)	57
2110	Embryonic shifting dunes	57
2120	Shifting dunes along the shoreline with Ammophila arenaria ("white dunes")	57
2130	Fixed coastal dunes with herbaceous vegetation ("grey dunes")	57
2140	Decalcified fixed dunes with Empetrum nigrum	57
2150	Atlantic decalcified fixed dunes (Calluno-Ulicetea)	57
2160	Dunes with Hippophaë rhamnoides	57
2170	Dunes with Salix repens ssp. argentea (Salicion arenariae)	57
2190	Humid dune slacks	57
21A0	Machairs (* in Ireland)	57
2250	Coastal dunes with Juniperus spp.	57
2330	Inland dunes with open Corynephorus and Agrostis grasslands	57
3110	Oligotrophic waters containing very few minerals of sandy plains (Littorelletalia uniflorae)	57
3130	Oligotrophic to mesotrophic standing waters with vegetation of the Littorelletea uniflorae and/or of the Isoëto-Nanojuncetea	57
3140	Hard oligo-mesotrophic waters with benthic vegetation of Chara spp.	57
3150	Natural eutrophic lakes with Magnopotamion or Hydrocharition - type vegetation	57

CODE	DESCRIPTION	PAGE NO
3160	Natural dystrophic lakes and ponds	57
3170	Mediterranean temporary ponds	57
3180	Turloughs	57
3260	Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation	57
4010	Northern Atlantic wet heaths with Erica tetralix	57
4020	Temperate Atlantic wet heaths with Erica ciliaris and Erica tetralix	57
4030	European dry heaths	57
4040	Dry Atlantic coastal heaths with Erica vagans	57
4060	Alpine and Boreal heaths	57
4080	Sub-Arctic Salix spp. scrub	57
5110	Stable xerothermophilous formations with Buxus sempervirens on rock slopes (Berberidion p.p.)	57
5130	Juniperus communis formations on heaths or calcareous grasslands	57
6130	Calaminarian grasslands of the Violetalia calaminariae	57
6150	Siliceous alpine and boreal grasslands	57
6170	Alpine and subalpine calcareous grasslands	57
6210	Semi-natural dry grasslands and scrubland facies on calcareous substrates (Festuco-Brometalia) (* important orchid sites)	57
6230	Species-rich Nardus grasslands, on silicious substrates in mountain areas (and submountain areas in Continental Europe)	57
6410	Molinia meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae)	57
6430	Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels	57
6510	Lowland hay meadows (Alopecurus pratensis, Sanguisorba officinalis)	57
6520	Mountain hay meadows	57
7110	Active raised bogs	57
7120	Degraded raised bogs still capable of natural regeneration	57
7130	Blanket bogs (* if active bog)	57
7140	Transition mires and quaking bogs	57
7150	Depressions on peat substrates of the Rhynchosporion	57
7210	Calcareous fens with Cladium mariscus and species of the Caricion davallianae	57
7220	Petrifying springs with tufa formation (Cratoneurion)	57
7230	Alkaline fens	57
7240	Alpine pioneer formations of the Caricion bicoloris-atrofuscae	57
8110	Siliceous scree of the montane to snow levels (Androsacetalia alpinae and Galeopsietalia ladani)	57
8120	Calcareous and calcshist screes of the montane to alpine levels (Thlaspietea rotundifolii)	57
8210	Calcareous rocky slopes with chasmophytic vegetation	57
8220	Siliceous rocky slopes with chasmophytic vegetation	57
8240	Limestone pavements	57
8310	Caves not open to the public	57
8330	Submerged or partially submerged sea caves	57
9120	Atlantic acidophilous beech forests with Ilex and sometimes also Taxus in the shrublayer (Quercion robori-petraeae or Ilici-Fagenion)	57
9130	Asperulo-Fagetum beech forests	57
9160	Sub-Atlantic and medio-European oak or oak-hornbeam forests of the Carpinion betuli	57
9180	Tilio-Acerion forests of slopes, screes and ravines	57
9190	Old acidophilous oak woods with Quercus robur on sandy plains	57
91A0	Old sessile oak woods with Ilex and Blechnum in the British Isles	57
91C0	Caledonian forest	57
91D0	Bog woodland	57
91E0	Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae)	57
91J0	Taxus baccata woods of the British Isles	57

3.1 Relative surface

CODE	DESCRIPTION	PAGE NO
A	> 15%-100%	58
B	> 2%-15%	58
C	≤ 2%	58

3.1 Degree of conservation

CODE	DESCRIPTION	PAGE NO
A	Excellent conservation	59
B	Good conservation	59
C	Average or reduced conservation	59

3.1 Global assessment

CODE	DESCRIPTION	PAGE NO
A	Excellent value	59
B	Good value	59
C	Significant value	59

3.2 Population (abbreviated to 'Pop.' in data form)

CODE	DESCRIPTION	PAGE NO
A	> 15%-100%	62
B	> 2%-15%	62
C	≤ 2%	62
D	Non-significant population	62

3.2 Degree of conservation (abbreviated to 'Con.' in data form)

CODE	DESCRIPTION	PAGE NO
A	Excellent conservation	63
B	Good conservation	63
C	Average or reduced conservation	63

3.2 Isolation (abbreviated to 'Iso.' in data form)

CODE	DESCRIPTION	PAGE NO
A	Population (almost) Isolated	63
B	Population not-isolated, but on margins of area of distribution	63
C	Population not-isolated within extended distribution range	63

3.2 Global assessment (abbreviated to 'Glo.' or 'G.' in data form)

CODE	DESCRIPTION	PAGE NO
A	Excellent value	63
B	Good value	63
C	Significant value	63

3.3 Assemblages types

CODE	DESCRIPTION	PAGE NO
WATR	Non-breeding waterbird assemblage	UK specific code
SBA	Breeding seabird assemblage	UK specific code
BBA	Breeding bird assemblage (applies only to sites classified pre 2000)	UK specific code

4.1 Habitat class code

CODE	DESCRIPTION	PAGE NO
N01	Marine areas, Sea inlets	65
N02	Tidal rivers, Estuaries, Mud flats, Sand flats, Lagoons (including saltwork basins)	65
N03	Salt marshes, Salt pastures, Salt steppes	65
N04	Coastal sand dunes, Sand beaches, Machair	65
N05	Shingle, Sea cliffs, Islets	65
N06	Inland water bodies (Standing water, Running water)	65
N07	Bogs, Marshes, Water fringed vegetation, Fens	65
N08	Heath, Scrub, Maquis and Garrigue, Phygrana	65
N09	Dry grassland, Steppes	65
N10	Humid grassland, Mesophile grassland	65
N11	Alpine and sub-Alpine grassland	65
N14	Improved grassland	65
N15	Other arable land	65
N16	Broad-leaved deciduous woodland	65
N17	Coniferous woodland	65
N19	Mixed woodland	65
N21	Non-forest areas cultivated with woody plants (including Orchards, groves, Vineyards, Dehesas)	65
N22	Inland rocks, Screes, Sands, Permanent Snow and ice	65
N23	Other land (including Towns, Villages, Roads, Waste places, Mines, Industrial sites)	65
N25	Grassland and scrub habitats (general)	65
N26	Woodland habitats (general)	65

4.3 Threats code

CODE	DESCRIPTION	PAGE NO
A01	Cultivation	65
A02	Modification of cultivation practices	65
A03	Mowing / cutting of grassland	65
A04	Grazing	65
A05	Livestock farming and animal breeding (without grazing)	65
A06	Annual and perennial non-timber crops	65
A07	Use of biocides, hormones and chemicals	65
A08	Fertilisation	65
A10	Restructuring agricultural land holding	65
A11	Agriculture activities not referred to above	65
B01	Forest planting on open ground	65
B02	Forest and Plantation management & use	65
B03	Forest exploitation without replanting or natural regrowth	65
B04	Use of biocides, hormones and chemicals (forestry)	65
B06	Grazing in forests/ woodland	65
B07	Forestry activities not referred to above	65
C01	Mining and quarrying	65
C02	Exploration and extraction of oil or gas	65
C03	Renewable abiotic energy use	65
D01	Roads, paths and railroads	65
D02	Utility and service lines	65
D03	Shipping lanes, ports, marine constructions	65
D04	Airports, flightpaths	65
D05	Improved access to site	65
E01	Urbanised areas, human habitation	65
E02	Industrial or commercial areas	65

CODE	DESCRIPTION	PAGE NO
E03	Discharges	65
E04	Structures, buildings in the landscape	65
E06	Other urbanisation, industrial and similar activities	65
F01	Marine and Freshwater Aquaculture	65
F02	Fishing and harvesting aquatic resources	65
F03	Hunting and collection of wild animals (terrestrial), including damage caused by game (excessive density), and taking/removal of terrestrial animals (including collection of insects, reptiles, amphibians, birds of prey, etc., trapping, poisoning, poaching, predator control, accidental capture (e.g. due to fishing gear), etc.)	65
F04	Taking / Removal of terrestrial plants, general	65
F05	Illegal taking/ removal of marine fauna	65
F06	Hunting, fishing or collecting activities not referred to above	65
G01	Outdoor sports and leisure activities, recreational activities	65
G02	Sport and leisure structures	65
G03	Interpretative centres	65
G04	Military use and civil unrest	65
G05	Other human intrusions and disturbances	65
H01	Pollution to surface waters (limnic & terrestrial, marine & brackish)	65
H02	Pollution to groundwater (point sources and diffuse sources)	65
H03	Marine water pollution	65
H04	Air pollution, air-borne pollutants	65
H05	Soil pollution and solid waste (excluding discharges)	65
H06	Excess energy	65
H07	Other forms of pollution	65
I01	Invasive non-native species	65
I02	Problematic native species	65
I03	Introduced genetic material, GMO	65
J01	Fire and fire suppression	65
J02	Human induced changes in hydraulic conditions	65
J03	Other ecosystem modifications	65
K01	Abiotic (slow) natural processes	65
K02	Biocenotic evolution, succession	65
K03	Interspecific faunal relations	65
K04	Interspecific floral relations	65
K05	Reduced fecundity/ genetic depression	65
L05	Collapse of terrain, landslide	65
L07	Storm, cyclone	65
L08	Inundation (natural processes)	65
L10	Other natural catastrophes	65
M01	Changes in abiotic conditions	65
M02	Changes in biotic conditions	65
U	Unknown threat or pressure	65
XO	Threats and pressures from outside the Member State	65

5.1 Designation type codes

CODE	DESCRIPTION	PAGE NO
UK00	No Protection Status	67
UK01	National Nature Reserve	67
UK04	Site of Special Scientific Interest (UK)	67
UK05	Marine Conservation Zone	67
UK06	Nature Conservation Marine Protected Area	67
UK86	Special Area (Channel Islands)	67
UK98	Area of Special Scientific Interest (NI)	67
IN00	Ramsar Convention site	67
IN08	Special Protection Area (SPA, EC Birds Directive)	67
IN09	Special Area of Conservation (SAC, EC Habitats Directive)	67

European Site Conservation Objectives for Outer Thames Special Protection Area Site Code: UK9020309

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A001 Gavia stellata; Red-throated diver (Non-breeding)

A193 Sterna hirundo; Common tern (Breeding)

A195 Sternula albifrons; Little tern (Breeding)

This is a European Marine Site

This SPA is a part of the Outer Thames European Marine Site (EMS). These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 20 December 2017 to reflect the consolidation of the Habitats Regulations in 2017.

Outer Thames Estuary SPA

Last updated: 13th September 2019

Supplementary advice

The Supplementary Advice on Conservation Objectives (SACOs) present attributes which are ecological characteristics or requirements of the classified species within a site. The listed attributes are considered to be those which best describe the site's ecological integrity and which if safeguarded will enable achievement of the Conservation Objectives.

Conservation Objectives relating to extent and distribution of habitat and population abundance are reflected in single attributes within the Supplementary Advice. Structure and function of habitats, and supporting processes for those habitats, are reflected in multiple attributes describing integrity of these ecological characteristics.

The Conservation Objective relating to the distribution of qualifying features (individual species or assemblages) may apply to most or all of the attributes listed in the SACOs and should be considered against them. Ensuring integrity of attributes relating to supporting habitats and processes should allow birds to distribute themselves optimally within (and, sometimes, outside) the SPA boundary. This is perhaps particularly relevant for food availability; extent and distribution of supporting habitat; quality of supporting habitat; predation; and disturbance caused by human activity.

Attributes have a target which is either quantified or qualified depending on the available evidence. The target identifies as far as possible the desired state to be achieved for the attribute. In many cases, the attribute targets show if the current objective is to either 'maintain' or 'restore' the attribute. The targets given for each attribute do not represent thresholds to assess the significance of any given impact in Habitats Regulation Assessments. You will need to assess this on a case-by-case basis using the most current information available.

Where there is no evidence to determine a marine feature's condition, a vulnerability assessment, which includes sensitivity and exposure information for features and activities in a site, has been used as a proxy for condition. Evidence used in preparing the SACO has been cited with hyperlinks included where possible. Where references have not been provided, Natural England has applied ecological knowledge and expert judgement.

Some, but not all, of these attributes can also be used for regular monitoring of the condition of the classified features. The attributes selected for monitoring the features, and the standards used to assess their condition, are listed in separate monitoring documents, which will be available from Natural England. As condition assessment information becomes available, the conservation advice package will be reviewed accordingly.

When to use

You should use this information, along with the conservation objectives and case-specific advice issued by Natural England when developing, proposing or assessing an activity, plan or project that may affect the site.

Any proposals or operations which may affect the site or its features should be designed so they do not adversely affect any of the attributes in the SACO or achievement of the conservation objectives.

Features:
Choose one or more features and/or their sub-features below by selecting the applicable boxes in the tree. This will show the relevant targets. Where a feature has sub-features this will be indicated with a greyed out triangle below, which can be expanded.

- ☐ Common tern (Sterna hirundo), Breeding
- ☐ Little tern (Sternula albifrons), Breeding
- ☐ Red-throated diver (Gavia stellata), Non-breeding

Reset | Select all | Show attributes and targets for selected features

Attributes:
You can filter to show only targets for certain attributes by selecting one or more attributes from the list below (use ctrl click to select multiple). Note that only attributes for the features you have chosen are shown.

Feature target

'Maintain' targets do not preclude the need for management, now or in the future, to avoid a significant risk of damage or deterioration to the feature. The supporting and/or explanatory notes in the SACOs set out why the target was chosen and any relevant site based supporting information. This is based on the best available information, including that gathered during monitoring of the feature's current condition.

Please select a feature.

See further guidance on how to undertake an HRA for a plan or project on a European site (https://www.gov.uk/guidance/habitats-regulations-assessments-protecting-a-european-site).

These tables bring together the findings of the best available scientific evidence which may be updated or supplemented in further publications from Natural England and other sources. You may decide to use other additional sources of information.

These tables do not give advice about SSSI features or other legally protected species which may also be present within the European site.

EC Directive 79/409 on the Conservation of Wild Birds

Citation for Special Protection Area (SPA)

Name: Sandlings

Unitary Authority/County: Suffolk

Consultation proposal: All or parts of Blaxhall Heath Site of Special Scientific Interest (SSSI), Leiston - Aldeburgh SSSI, Sandlings Forest SSSI, Snape Warren SSSI, Sutton & Hollesley Heaths SSSI and Tunstall Common SSSI have been recommended as a Special Protection Area because of their European ornithological importance. In particular, for their breeding populations of Nightjars *Caprimulgus europaeus* and Woodlarks *Lullula arborea*.

Site description: The Sandlings SPA lies near the Suffolk Coast between the Deben Estuary and Leiston. In the 19th century, the area was dominated by heathland developed on glacial sandy soils. During the 20th century, large areas of heath were planted with blocks of commercial conifer forest and others were converted to arable agriculture. Lack of traditional management has resulted in the remnant areas of heath being subject to successional changes, with the consequent spread of bracken, shrubs and trees, although recent conservation management work is resulting in their restoration. The heaths support both acid grassland and heather-dominated plant communities, with dependant invertebrate and bird communities of conservation value. Woodlark *Lullula arborea* and Nightjar *Caprimulgus europaeus* have also adapted to breeding in the large conifer forest blocks, using areas that have recently been felled and recent plantation, as well as areas managed as open ground.

Size of SPA: The SPA covers an area of 3,391.80 ha.

Qualifying species:
The site qualifies under **article 4.1** of the Directive (79/409/EEC) as it is used regularly by 1% or more of the Great Britain populations of the following species listed in Annex I in any season:

Annex 1 species	Count and Season	Period	% of GB population
Nightjar *Caprimulgus europaeus*	109 males - breeding	Count as a 1992	3.2% GB
Woodlark *Lullula arborea*	154 pairs - breeding	Count as at 1997	10.3% GB

Bird figures from:

Morris, A., Burges, D., Fuller, R.J., Evans, A.D. & Smith, K.W. 1994. The status and distribution of nightjars *Caprimulgus europaeus* in Britain in 1992. A report to the British Trust for Ornithology. *Bird Study* **41**: 181-191.

Wotton, S.R. & Gillings, S. 2000. The status of breeding woodlarks in Britain in 1997. *Bird Study* **47**: 212-224.

Status of SPA
Sandlings was classified as a Special Protection Area on 10 August 2001.

European Site Conservation Objectives for Sandlings Special Protection Area Site Code: UK9020286

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A224 *Caprimulgus europaeus*; European nightjar (Breeding)

A246 *Lullula arborea*; Woodlark (Breeding)

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

COUNTY: SUFFOLK SITE NAME: SIZEWELL MARSHES

DISTRICT: SUFFOLK COASTAL

Status: Site of Special Scientific Interest (SSSI) notified under Section 28 of the Wildlife and Countryside Act 1981 as amended

Local Planning Authority: SUFFOLK COUNTY COUNCIL, Suffolk Coastal District Council

National Grid Reference: TM 466638 Area: 104.33 (ha.) 257.80 (ac.)

Ordnance Survey Sheet 1:50,000: 156 1:10,000: TM 46 SE

Data Notified (Under 1949 Act): – Date of Last Revision: –

Date Notified (Under 1981 Act): 1987 Date of Last Revision: 1992

Other Information:
The site has been extended at the 1992 revision.

Description and Reasons for Notification:
Sizewell Marshes are important for their large area of lowland, unimproved wet meadows which support outstanding assemblages of invertebrates and breeding birds. Several nationally scarce plants are also present.

The site occupies a low-laying basin of deep fen peat. The water table is permanently high, with the area being prone to flooding, and there is an extensive network of ditches across the site.

In the areas of unimproved wet meadow the principal grass species are Sweet Vernal-grass *Anthoxanthum odoratum*, Crested Dog's-tail *Cynosurus cristatus*, Rough-stalked Meadow-grass *Poa trivialis* and Yorkshire-fog *Holcus lanatus*. There are many other typical species including Marsh Pennywort *Hydrocotyle vulgaris*, Ragged Robin *Lychnis flos-cuculi*, Large Bird's-foot-trefoil *Lotus uliginosus*, Marsh-orchids *Dactylorhiza* spp., Bogbean *Menyanthes trifoliata*, Bog Pimpernel *Anagallis tenella*, Yellow Iris *Iris pseudacorus*, sedges *Carex spp*. and rushes *Juncus spp*. The nationally scarce Marsh Dock *Rumex palustris* and Greater Water-parsnip *Sium latifolium* are also present. It is considered that these communities are representative of the *Juncus subnodulosus – Cirsium palustre* fen-meadow and the *J. effusus/acutiflorus – Galium palustre* rush-pasture, as described in the National Vegetation Classification. In addition, several areas of reedbed dominated by Common Reed *Phragmites australis* and alder carr occur.

The extensive ditch system supports a diverse aquatic flora which includes the nationally scarce Soft Hornwort *Ceratophyllum submersum*, Fen Pondweed *Potamogeton coloratus* and Whorled Water-milfoil *Myriophyllum verticillatum*. The variety of ditch depths and widths, together with their fringing vegetation provide an important contribution to the site's habitat value for invertebrates and birdlife.

Sizewell Marshes are of exceptional interest for their invertebrate fauna, supporting a wide range of taxa and many nationally rare or scarce species. These include terrestrial and aquatic beetles (Coleoptera), flies (Diptera), moths (Lepidoptera), dragonflies (Odonata) and spiders (Araneae).

The breeding bird assemblage is also of national significance with many species that are typical of wet grassland and associated habitats, including Shoveler, Gadwall, Teal, Snipe and Lapwing.

Operations likely to damage the special interest

Site name: Sizewell Marshes, Suffolk

OLD1003416

Ref. No.	Type of Operation
1	Cultivation, including ploughing, rotovating, harrowing, and re-seeding.
2	Grazing and changes in the grazing regime (including type of stock or intensity or seasonal pattern of grazing and cessation of grazing).
3	The introduction of stock feeding and changes in stock feeding practice.
4	The introduction of mowing and changes in the mowing or cutting regime (including hay making to silage and cessation).
5	Application of manure, fertilisers and lime.
6	Application of pesticides, including herbicides (weedkillers).
7	Dumping, spreading or discharge of any materials.
8	Burning.
9	The release into the site of any wild, feral or domestic animal*, plant or seed.
10	The killing or removal of any wild animal*, including pest control.
11	The destruction, displacement, removal or cutting of any plant or plant remains.
12	The introduction of tree and/or woodland management+ and changes in tree and/or woodland management+.
13a	Drainage (including the use of mole, tile, tunnel or other artificial drains).
13b	Modification of the structure of watercourses (eg streams, springs, ditches, dykes, drains), including their banks and beds, as by re-alignment, re-grading and dredging.
13c	Management of aquatic and bank vegetation for drainage purposes (see also 11).
14	The changing of water levels and tables and water utilisation (including irrigation, storage and abstraction from existing water bodies and through boreholes).
15	Infilling of ditches, dykes, drains and marshes.
16a	The introduction of freshwater fishery production and/or management** and changes in freshwater fishery production and/or management.
20	Extraction of minerals, including peat, shingle, sand and gravel, topsoil, subsoil, and spoil.
21	Construction, removal or destruction of roads, tracks, fences, banks ditches or other earthworks, or the laying, maintenance or removal of pipelines and cables, above or below ground.
22	Storage of materials.
23	Erection of permanent or temporary structures, or the undertaking of engineering works, including drilling.
26	Use of vehicles or craft likely to damage or disturb features of interest.
27	Recreational or other activities likely to damage features of interest.
28	Game and waterfowl management and hunting practices and changes in game and waterfowl management and hunting practice.

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* ‘animal’ includes any mammal, reptile, amphibian, bird, fish or invertebrate.

\+ including afforestation, planting, clear and selective felling, thinning, coppicing, modification of the stand or underwood, changes in species composition, cessation of management.

** including sporting fishing and angling.

Views About Management

ENGLISH NATURE

A statement of English Nature's views about the management of Sizewell Marshes Site of Special Scientific Interest (SSSI).

This statement represents English Nature's views about the management of the SSSI for nature conservation. This statement sets out, in principle, our views on how the site's special conservation interest can be conserved and enhanced. English Nature has a duty to notify the owners and occupiers of the SSSI of its views about the management of the land.

Not all of the management principles will be equally appropriate to all parts of the SSSI. Also, there may be other management activities, additional to our current views, which can be beneficial to the conservation and enhancement of the features of interest.

The management views set out below do not constitute consent for any operation. English Nature's written consent is still required before carrying out any operation likely to damage the features of special interest (see your SSSI notification papers for a list of these operations). English Nature welcomes consultation with owners, occupiers and users of the SSSI to ensure that the management of this site conserves and enhances the features of interest, and to ensure that all necessary prior consents are obtained.

Management Principles

Ditches

Ditches are artificial habitats created by land drainage, or occasionally by the channelisation of small streams. They may represent the only remaining freshwater habitat within former wetland areas, and often support a wide range of aquatic plant and animal (in particular invertebrate) species that would have previously been more widespread in ponds and wetlands.

If left unmanaged, silt accumulates in the bottom of the ditches, and emergent plants such as reeds are able to colonise across the width of the ditch, leading to a loss of aquatic plant diversity and a gradual drying out of the ditch. To prevent this, periodical removal of sediment and vegetation may be necessary to return the ditch to an early stage of the management cycle. Ideally, ditch management should be undertaken on a rotation, creating a series of different management stages across a site at any one time. All stages of the management cycle have wildlife interest; recently cleared ditches are good for plants and animals which favour newly created habitats and cannot tolerate competition with other species; middle-stage ditches support a rich aquatic plant flora; and late-stage ditches may be important for a variety of invertebrates. The removal of both sediment and vegetation is usually better than simply cutting the vegetation, which does not recreate the earliest stages of the ditch

management cycle. Where possible, management should aim to create shallow shelving margins rather than steep ditch sides. Where water voles are known to be present, the relevant good practice guidelines for ditch management and conservation should be followed.

Most ditch systems are subject to water level control, which should be managed to ensure that there is a sufficient depth of water (0.3-0.5m) in ditches throughout the year. Rapid or extreme changes in water level should be avoided unless they are known to be important to plant or animal communities relying on such fluctuations.

Ditches are susceptible to increased levels of nutrients which can cause a loss of aquatic plants and increases in algal growth. Other activities that can lead to this include the control or removal of aquatic plants, or the introduction of species such as bottom feeding coarse fish which uproot plants and disturb ditch sediments. Ditches are also susceptible to invasion by non-native aquatic plants such as floating pennywort and water fern, which are able to grow rapidly taking up available habitat and smothering other plants. Some native plants including a number of duckweed species are also able to take over in this way (although such growths are usually exacerbated by increased nutrients in the water) and management may be necessary to control such invasions where they cause a problem.

Wet grassland with breeding and wintering bird interest

Wet grasslands occur on land that is subject to periodic flooding or has a seasonally high water table and is waterlogged for much of the year. Wet grassland often supports a wide variety of plants and animals, in particular birds and invertebrates, and is an important habitat for breeding waders and wintering wildfowl.

Wet grassland requires active management if it is to retain its conservation interest. Generally, each year's growth of vegetation must be removed. Otherwise the sward becomes dominated by tall, vigorous grasses and rushes which, together with an associated build up of dead plant matter, suppress less vigorous species and lower the botanical richness of the sward. Traditionally, this management is achieved by grazing. Cattle are often the preferred stock, being relatively tolerant of wet conditions and able to control tall grasses and rank vegetation. Cattle also tend to produce a rather uneven, structurally diverse sward. However, ponies, or even hill sheep, can be used if necessary. Grazing usually takes place at times between late spring and early autumn, but the precise timing and intensity will depend on local conditions and requirements, such as the need to avoid trampling ground-nesting birds. Heavy poaching should be avoided but light trampling can be beneficial in breaking down leaf litter and providing areas for seed germination. Agricultural operations in general should be avoided before mid-June to minimise disturbance to breeding birds or the destruction of nests. An element of managed scrub, both within and fringing a field can be of importance to birds and invertebrates, as can a surrounding hedge.

Partial winter flooding is important in maintaining suitable habitat conditions for wintering birds. A mosaic of winter flooded grassland and permanently un-flooded grassland is desirable, with both temporary and permanent pools present. The maintenance of a mosaic of shallow surface pools and un-flooded areas during the

winter will provide roosting and feeding habitat for wintering wildfowl and waders. From April onwards, the area of standing surface water should be reduced to increase the area available for nesting waders and also by concentrating aquatic invertebrates in small pools to provide suitable feeding areas for their young. Some shallow areas of flooding should be maintained until late June to provide patches of bare muddy ground on which the birds and their young can feed as raised sward height makes feeding on the drier areas more difficult. The birds using these features are directly vulnerable to disturbance, which can cause them to lose time spent feeding or drive them to areas with a poorer supply of food. Management should seek to minimise any harmful disturbance, especially at times when bird populations are under stress, such as severely cold conditions. Predators, especially crows and related species, should be controlled and this may be best achieved by limiting their nesting sites.

All habitats

The habitats within this site are highly sensitive to inorganic fertilisers and pesticides, applications of which should be avoided both within the site itself and in adjacent surrounding areas. Herbicides may be useful in targeting certain invasive species, but should be used with extreme care. Access to this site, and any recreational activities within, may also need to be controlled.

Appendix 4: References to Written Submission

for the Royal Society for the Protection of Birds and Suffolk Wildlife Trust

Submitted for Deadline 2
2 June 2021

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Planning Inspectorate Ref: EN010012
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HABITAT USE AND SELECTION OF THE MARSH HARRIER *CIRCUS AERUGINOSUS* IN AN AGRICULTURAL-WETLAND MOSAIC

USO Y SELECCIÓN DE HÁBITAT DEL AGUILUCHO LAGUNERO *CIRCUS AERUGINOSUS* EN UN MOSAICO DE AGRICULTURA Y HUMEDAL

Michelle Alves[1, 2], Joaquim Pedro Ferreira[1], Inês Torres[1], Carlos Fonseca[1] and Milena Matos[1] *

Summary.—Studying various aspects of the biology of a species of conservation concern allows us to improve our understanding of, for example, how the particularities of the landscape matrix influence its occurrence. The Lower Vouga Lagoon in central-western Portugal offers an unusual diversity of natural and humanised biotopes in an agricultural-wetland complex mosaic. This study aimed to identify the factors that may influence the occurrence and abundance of a diurnal raptor, the marsh harrier *Circus aeruginosus*, in this kind of landscape. The marsh harrier is a top predator that is considered as an umbrella species, i.e. one whose conservation may also help preserve and promote biodiversity in general in its community. This study provides potentially valuable information on management and conservation strategies in both natural and agricultural areas. We found that natural habitats, such as reedbeds, are a key habitat for this predator, providing shelter, food and suitable nesting sites. Despite the general negative effect of human pressure on the occurrence of marsh harriers, the species seems to tolerate and even benefit from humanised environments, such as rice fields, which constituted the preferred foraging habitat during the non-breeding period. Nevertheless, the degree of disturbance should be carefully considered, since road density and agricultural machinery negatively influenced the presence of the species in the landscape, particularly during the breeding period. The preservation of the distinctive features of the studied mosaic is very important. It is also necessary to establish a conservation programme involving the local human community.

Key words: conservation, humanised landscape, Lower Vouga Lagoon, reed beds, spatial heterogeneity.

Resumen.—El estudio de varios aspectos de la biología de una especie con problemas de conservación nos permite comprender mejor cómo, por ejemplo, las particularidades del hábitat pueden influir en la distribución de esa especie. El Baixo Vouga Lagunar, centro-oeste de Portugal, ofrece una diversidad inusual de biotopos naturales y humanizados, en un complejo mosaico de agricultura y humedal. El objetivo de este estudio es comprender qué factores pueden estar influenciando la presencia y abundancia de una rapaz diurna, el aguilucho lagunero *Circus aeruginosus*, en este particular mosaico. El aguilucho lagunero es un predador considerado una especie paraguas, esto es, una especie cuya conservación puede también ayudar a preservar y promover la biodiversidad en general. Este estudio propor-

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ciona información potencialmente valiosa para la gestión y las estrategias de conservación en las zonas naturales y agrícolas. Encontramos que los hábitats naturales, como los carrizales, son un hábitat clave para este predador, proporcionando refugio, alimentos y lugares de nidificación adecuados. A pesar del efecto negativo general de la presión humana sobre la presencia de aguilucho lagunero, la especie mostró que también puede tolerar, e incluso beneficiarse, de los ambientes humanizados, como los arrozales, que constituyen el hábitat preferido de alimentación durante el período no reproductivo. Sin embargo, el grado de perturbación debe ser considerado cuidadosamente, ya que la densidad de carreteras y máquinas agrícolas influyen negativamente en la presencia de la especie, sobre todo durante la época de cría. La preservación de las características distintivas del paisaje estudiado debería ser prioritaria, así como el establecimiento de un programa de conservación para ser llevado a cabo a nivel local.

Palabras clave: Baixo Vouga Lagunar, conservación, carrizales, heterogeneidad espacial, paisaje humanizado.

Introduction

The presence of animals in specific landscapes indicates that species have priorities, preferring some places over others, in order to establish their populations (Alcock and Farley, 2009), which leads species to be more abundant in certain environments than in others (Morris, 1987). In nature, top predators such as raptors play a key role in the resilience of biodiversity, determining the patterns of the composition of the ecosystem (Terborgh, 1988; Luo *et al*., 2010).

The concept of habitat selection implies a preference of a species for a specific habitat type, whereas recording habitat use only notes the presence/occupation of the species without considering habitat availability within the landscape (Sutherland and Green, 2008). Habitat use and selection can be influenced by several factors, such as vegetation structure and composition, food resources, human disturbance, microclimate and biotic factors (e.g. presence of predators or conspecifics), making these important features to consider when evaluating habitat quality for a particular species (Cardador *et al*., 2012a; Rotenberry, 1985; Firkowski, 2006). In particular, habitat use and selection within humanised landscapes are important aspects to consider in wildlife studies, since habitats provide food, shelter and breeding sites, which are essential for species survival (Johnson, 1980; Morris, 1987; Firkowski, 2006). Changes in landscapes can induce habitat loss and fragmentation, by reducing the size of optimal habitat patches and increasing the distance between them respectively, and may contribute to the development of new environments. As these new conditions may or may not be usable by animals (Andren, 1994; Williams *et al*., 2013), the species composition within communities may potentially be changed. Human activities strongly affect the landscape, generally reducing the amount and the quality of available habitat for many species (Murcia, 1995). The transition from natural habitats to anthropogenic habitats such as monocultural exotic forests, farmlands or pastures, are examples of some severe landscape changes (Saunders *et al*., 1991; Herkert, 1994).

The marsh harrier *Circus aeruginosus* is a medium-sized raptor that occurs in a wide range of open habitats, including agricultural landscapes (Cardador *et al*., 2001; Cardador and Mañosa, 2011), and requires large areas of land to hunt and breed (Cramp and Simmons, 1983). In Portugal, it is a resident species that occurs mostly in the western coastal strip, especially in wetlands and in some agricultural areas (Cabral *et al*., 2005). In recent decades the species has undergone a moderate increase across Europe (Cramp and Simmons, 1994; BirdLife International,

2004; Cardador *et al*., 2012b). However, in Portugal, the increase in the abundance was not accompanied by an expansion in its distribution area, and the species is still classified as 'Vulnerable' (Cabral *et al*., 2005). The main threats to its survival or reproduction include habitat loss, degradation of wetlands, urbanisation and other disturbances caused by human activities (Del Hoyo *et al*., 1994; Fernandes *et al*., 1996). In some wetlands, water pollution with heavy metals and pesticides from industrial and agricultural effluents is another pertinent factor (Pedrocchi, 1994; Rodrigues, 1998). However, with respect to agriculture, recent studies demonstrate that the species seems to tolerate a certain degree of intensification and may even benefit from it (Cardador *et al*., 2011; Sternalski *et al*., 2013). The marsh harrier has a preference for marshland vegetation, especially reed beds of *Phragmites australis*, where it forms winter dormitories, nests and hunts. Reeds provide good thermal conditions and reduce the effects of human disturbance and of predation by terrestrial mammals (Costa *et al*., 1993; Rosa *et al*., 2001; Oliver, 2005; Verma and Prakash, 2007), these latter factors being main causes of reproductive failure (Underhill-Day, 1984).

Given that high quality habitats, offering feeding, nesting and resting opportunities, are essential for the maintenance of the species, understanding the patterns of occurrence of the marsh harrier within human-transformed mosaics is of vital importance for the success of any attempt to reconcile development and conservation. The aim of this study was to assess the habitat use and selection of the marsh harrier in a complex agricultural-wetland mosaic, defining the most important aspects of its spatial ecology. Despite the conservation status of this species, the threats to its survival and its ecological importance, it remains relatively poorly studied among raptor species, especially in Portugal. Thus, we believe that this study will add valuable knowledge about its ecology. We also aim at contributing to the general conservation of the biodiversity of wetlands that are under human pressure.

Material and methods

Study Area

The Lower Vouga Lagoon is included in the Vouga River Estuary, located in the District of Aveiro, central-western Portugal, approximately between 8º 32' 57" W and 8º 41' 32" W; and 40º 49' 43" N and 40º 41' 32" N (fig. 1). The region presents characteristics of the Mediterranean climate with strong Atlantic influences, with an annual mean temperature of about 15ºC. The average annual humidity is 77% and the average annual rainfall is 1,387 mm, limited rainfall occurring in the summer. This agricultural-wetland complex occupies approximately 12,205 ha, and encompasses important ecosystems integrated in the Natura 2000 Network (Special Protection Area of Ria de Aveiro, PTZPE0004, and Site of Community Importance, PTCON006), featuring one of the most important Portuguese wetlands. The system integrates a variety of natural and humanised habitats, such as pastures (2.35% of the area), fields of rice *Oryza sativa* (0.98%) and maize *Zea mays* (28.13%); and "Bocage" (7.09%), a typical and rare landscape type that consists of hedgerows of trees (e.g. *Salix alba*), shrubs (e.g. *Rubus ulmifolius*) and ditches that compartmentalise farmlands and pastures. The wetlands are composed of reed beds of *Phragmites australis* (4.50%), saltmarshes of *Spartina maritima* (12.79%) and rush beds of *Juncus maritimus* (6.67%) (Alves, 2013; Pinho *et al*., 2003). This region also houses high faunal richness, from all taxonomical groups. According to national bird censuses, the study area shelters 11 to 12 breeding resident pairs of marsh harriers, some 17% of the national breeding population (Rosa *et al*., 2006). The number of marsh harriers in the study area almost doubles in autumn and winter in some years (IDAD, 2008).

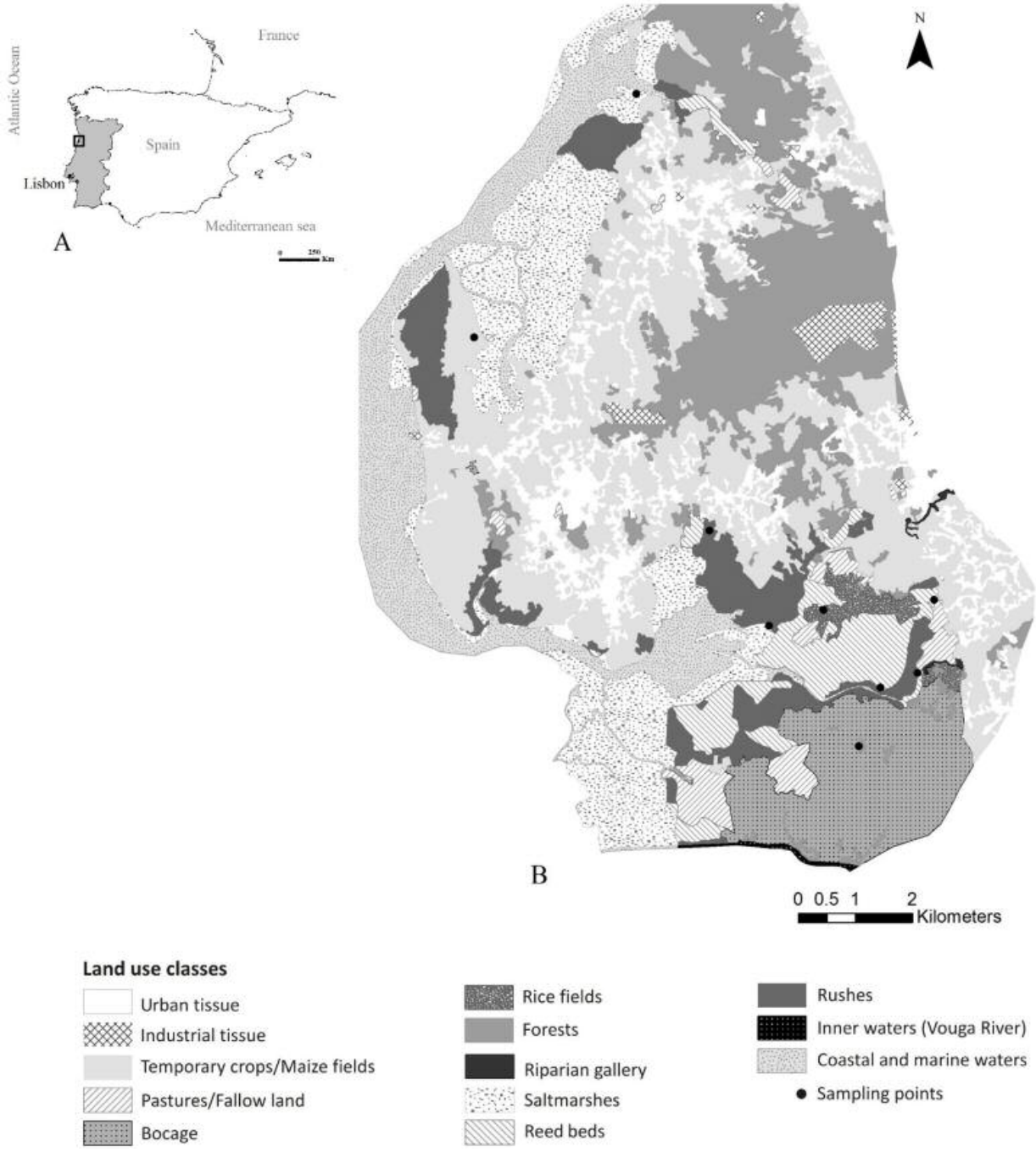


FIG. 1.—Location of the study area in the Iberian Peninsula and in Portugal (A). Distribution of land use within the study area, with sampling points (B).
[Situación del área de estudio en la península Ibérica y en Portugal (A). Cobertura del suelo en el área de estudio, con los puntos de muestreo (B).]

Habitat use and selection

Fieldwork was conducted using direct observation of birds from November 2011 to October 2012. A set of 25 random areas, at least 1 km apart, was generated in ArcGIS 9.3®ESRI. After field prospection, we selected nine fixed observation points based on: the existence of high observation points such as fire watch-towers or landscape observatories,

their reasonable distance from urban areas and their accessibility. The land cover of the entire study area was mapped and characterised through a Geographic Information System (GIS). The habitats considered in the study were reed beds, saltmarshes, rushes, "Bocage", rice fields and maize fields.

The frequency and duration of sampling sessions are presented in table 1. April 2012 was only sampled once due to heavy rain. Sampling was always performed by the same observer (MA), from each point in turn. Throughout the year, efforts were made to alternate the sampling hour at each observation point, in order to reduce potential bias related to daily activity patterns. During each sampling session, an area of about 1.5 km radius was covered using binoculars and telescope. This radius delimits the sampling area around the observation points that allowed the correct identification of the species and habitat. Each time a marsh harrier was observed, the displayed behaviour -such as flight, mating, nest building, hunting and resting– was recorded, as well as the habitat over which it took place. For each sampling session, weather conditions –temperature, relative humidity, wind speed, visibility– were measured using a portable weather station (Kestrel® 4000). Relevant human activities in the landscape, such as cropping, agricultural machinery working at the site or the presence/absence of livestock, were also recorded.

Statistical analyses

Habitat use and selection

The evaluation of the habitat use was based on the number of observations recorded in each habitat type. For the analysis, we considered

TABLE 1

Frequency and duration of sampling sessions per month.
[Frecuencia y duración de las sesiones de muestreo por mes.]

Month	Nr sampling points	Nr sampling sessions per month and per point	Duration of each sampling session (hours)	Total observation hours per month and per point
Nov 2011	9	1	1	1
Dec 2011	9	1	1	1
Jan 2012	9	2	1	2
Feb 2012	9	2	2	4
Mar 2012	9	2	2	4
Apr 2012	9	1	2	2
May 2012	9	2	2	4
Jun 2012	9	2	2	4
Jul 2012	9	2	2	4
Aug 2012	9	2	2	4
Sep 2012	9	2	2	4
Oct 2012	9	1	2	2
Total observation hours per point				**36**

three different types of behaviours, namely 'presence' (birds flying or resting), 'foraging' (confirmed hunting behavior and successful hunting attempts) and 'reproduction' (all activities related to reproduction, from courtship and nest building to chick feeding). Observation data were also seasonally grouped (under breeding and non-breeding periods) and were standardised for sampling effort. To assess habitat selection we used the Savage selectivity index (e.g. Tella and Forero, 2000; Ursúa *et al*., 2005; Catry *et al*., 2013): $wi = Ui/pi$, where Ui is the proportion of observations recorded in a given habitat and pi is the proportion of that habitat related to the total habitat available. This index varies from 0 (maximum negative selection) to infinity (maximum positive selection), 1 indicating no selection (Manly *et al*., 1993). The statistical significance of this index was obtained by comparing the statistic $(wi - 1)^2/\mathrm{se}(wi)^2$ with the corresponding critical value of a Chi-square distribution with one degree of freedom. The standard error $(\mathrm{se}(wi))$ was calculated using the formula $\sqrt{[(1 - pi)/(u_+ * pi)]}$, where u_+ is the total number of observations in a given habitat. All comparisons were corrected for multiple tests using the Bonferroni criteria.

Presence and abundance of marsh harriers

We modelled two dependent variables: presence (presence/absence data) and abundance (number of individuals) of marsh harriers during the breeding and non-breeding periods. We grouped the independent variables into categories related to: 1) landscape, 2) food availability, 3) abiotic factors and 4) human disturbance (table 2). Landscape variables included six classes of land cover, corresponding to the habitats considered in this study (reed beds, saltmarshes, rushes, "Bocage", rice fields and maize fields). We measured the length of water ditches, as well as their occupancy within the considered sampling areas (each sampling area was divided into four quadrants and those with ditches were counted) from our GIS database. We measured seasonal changes within the landscape in terms of the number of times each habitat experienced a structural alteration, *e.g*. from cultivated to fallow land or vice versa. We calculated spatial heterogeneity using the Shannon diversity index (Magurran, 2004), based on the area of each habitat within the considered sampling areas.

Studies of small mammal communities and the diet of marsh harriers were also conducted in the study area during fieldwork (Alves, 2013). Small mammals were the most important prey, accounting for 68% frequency of occurrence in the diet. Thus, food availability for this study consisted of the field abundance of the four most important prey species represented within the diet (small mammal species *Apodemus sylvaticus*, *Crocidura russula*, *Mus* spp., *Microtus* spp.), in the habitats of the respective sampling area.

Abiotic factors were tested using the meteorological variables measured during sampling sessions. Variables related to human disturbance were established based on the extent of constructions (e.g. buildings, industry), on the presence of livestock and agricultural machinery working during sampling sessions, and on the length of roads within each sampling area. As with ditches, the occupancy of roads and constructions was also evaluated according to their presence in the quadrants of the sampling area.

We also included 'behaviour' as a factor to control variability in models, to avoid some overrepresented behaviors affecting the results. While foraging was included in breeding and non-breeding periods, by definition reproduction was only considered in the breeding period.

Generalised linear mixed models (GLMM) were used to identify the factors determining the presence and abundance of marsh harriers. We considered sampling points as random factors for each observation data. We analyzed the presence of marsh harriers with a binomial distribution and a logit link function and the

TABLE 2

Variables considered in each category, their respective code, and value range assigned to each category. NB: non-breeding period and B: breeding period – "x" indicates the variables that were used in each analysis, after removing the correlated variables.
[Variables consideradas en cada categoría, sus códigos respectivos y rango de valores asignados a cada categoría. NB: periodo sin reproducción, B: periodo de reproducción – "x" indica qué variables se usaron en cada análisis, tras eliminar variables correlacionadas.]

Variable types	Code	Range or Categorical value	Presence/Abundance			
			NB	B	NB	B
Behaviour						
Foraging	Forag	0 – 1	x	x	x	x
Reproduction	Reprod	0 – 1		x		x
Landscape						
Waterlines (extension)	Line_W	5.2 – 22.59 km				
Waterlines (occupancy within sampling area)	QW	0 – 4				
Spatial heterogeneity	Spa_het	1.07890 – 1.97240 H'	x		x	x
Landscape alteration within the period	Chan_land	0 – 6	x	x	x	
Rice fields	Rice	0 – 91.55 ha				
Bocage	Boc	0 – 271.69 ha				
Reedbeds	Reed	0 – 274.01 ha	x	x	x	x
Maize fields	Maize	0 – 325.56 ha		x		
Rushes	Rus	37.10 – 210.94 ha				
Saltmarshes	Sal	0 – 77,71 ha	x	x	x	x
Food availability						
Apodemus sylvaticus	Apo	0 – 20.78*	x	x	x	x
Crocidura spp.	Cro	0 – 3.92*	x	x	x	x
Microtus spp.	Mic	0 – 2.83*	x	x	x	x
Mus spp.	Mus	0 – 12.52*	x	x	x	x
Abiotic factors						
Relative humidity	Rhum	13 – 94 %	x	x	x	x
Atmospheric pressure	Atm	1000.70 –1055.20 hPa	x	x	x	x
Air temperature	Temp	5.10 – 32 ºC	x	x	x	x
Wind speed	Ws	0 – 6.70 m/s	x	x	x	x
Human disturbance						
Presence of cattle	Cattle	0 – 1	x	x	x	x
Human constructions	Const_Hum	0 – 1.13 km^2			x	x
Roads	Road	15.82 – 45.72 km	x	x	x	x
Presence of agricultural machinery	Agr_Mac	0 – 1	x	x	x	x
Roads (occupancy within sampling area)	QR	0 – 4				
Human constructions (occupancy within sampling area)	QH	0 – 4	x	x		

** Small mammal index of abundance: number of captures per 100 trap–nights.*

abundance data with a Poisson distribution with a log link. We used the R software (R Core Team, 2013), function 'glmer', package 'lme4' (Bates *et al*., 2011).

We assessed the correlation between independent variables with Spearman's Correlation Coefficient (Sokal and Rohlf, 1995). If two variables were highly correlated ($r_s > 0.70$) only one data set was selected. The variable showing the lowest correlation with the dependent variable or, under special circumstances, the variable with less justified biological interpretation, was eliminated (Mladenoff *et al*., 1999). We used the Akaike Information Criterion (AIC) to rank models according to their capacity to describe the data parsimoniously (Burnham and Anderson, 2002).

Results

In total, 324 hours of observations were conducted during 180 sampling sessions throughout an annual cycle (table 1), and 267 observations of marsh harriers were recorded. Marsh harriers were observed in 85% of the sampling sessions. Field observations and previous censuses (Rosa *et al*., 2006) agree on an estimate of 11 to 12 resident breeding pairs in the study area.

Habitat use and selection

We observed that marsh harriers positively selected reed beds, saltmarshes and rice fields, during both the breeding and non-breeding periods (table 3). Whereas they preferred reed beds for foraging during the breeding season, they selected both reed beds and rice fields during the non-breeding period. Maize fields were avoided during the breeding period and never used for foraging during the non-breeding period, despite being the most extensive habitat in the study area. No foraging activities were observed in Bocage. Reproductive behaviours were observed solely in reed beds. Rushes showed consistent negative selection throughout the year (table 3).

Presence and abundance of marsh harriers

The best model describing the presence of marsh harriers in the Lower Vouga Lagoon, during the breeding period (table 4) includes variables related to landscape, food availability and human presence. Spatial heterogeneity, reed beds, saltmarshes and maize fields presented a significant positive association with the species presence, whereas roads and human constructions were significant and negatively associated with the species presence (table 5). During the non-breeding period, the best model describing the species presence included variables related to landscape, food availability, abiotic factors and human presence (table 4). The most significant variables were reed beds and roads, with a positive influence, and the abundance of *Crocidura russula* and agriculture machinery with a negative influence (table 5).

Reed beds were the only variable explaining the abundance of marsh harriers during the breeding period, with a significant positive influence (tables 4 and 5). In the non-breeding period, the species abundance was best explained by the model including reed beds, food availability (*Mus* spp.) and human presence (table 4), with reed beds presenting the most significant positive influence and road occupancy presenting the most significant negative influence (table 5). Foraging behaviour was strongly associated with the abundance of marsh harriers during the non-breeding season (tables 4 and 5).

Discussion

Habitat use and selection

In general, the pattern of use and selection of habitats can be explained by two important factors: provision of a suitable nesting location

TABLE 3

Proportion of habitat available and used by marsh harriers (presence, foraging and nesting) during breeding and non-breeding periods, Savage Selectivity Index (w*i*) of the different habitats in those periods, and standard error (s.e.). Threshold for significance was set at 0.05; 0.008 and 0.0125 for presence and foraging, respectively, in both breeding and non-breeding periods, after correction for multiple comparisons. NS: non-significant; *: significant. Habitat availability in sampling areas (%) = rice fields (6.05), Bocage (18.39), reed beds (20.82), rushes (18.35), maize fields (23.87) and saltmarshes (12.50).
*[Proporción de hábitat disponible y utilizado por los aguiluchos laguneros (presencia, alimentación y nidificación) durante los periodos de reproducción y sin reproducción, Índice de Selectividad de Savage (w*i*) de los diferentes hábitats en esos periodos y error estándar (s.e.). El nivel de significación se estableció en 0,05; 0,008 y 0,0125 para presencia y alimentación, respectivamente, tanto durante el periodo de reproducción como sin reproducción, después de corrección a causa de comparaciones múltiples. NS: no significativo; *: significativo. Disponibilidad de hábitats en áreas muestreadas (%) = arrozales (6,05), Bocage (18,39), carrizales (20,82), juncales (18,35), maizales (23,87) y marismas salobres (12,50).]*

Habitat	Use (%)	w*i*	s.e.	'Statistic'	P
BREEDING SEASON					
Presence					
Rice fields	7.20	1.19	0.09	4.56	NS
Bocage	12.00	0.65	0.09	13.54	NS
Reed beds	78.43	3.77	0.09	851.94	*
Rushes	8.00	0.43	0.09	35.68	*
Maize fields	3.38	0.14	0.10	81.56	*
Saltmarshes	28.29	2.26	0.09	183.61	*
Foraging					
Rice fields	0	–	–	–	–
Bocage	0	–	–	–	–
Reed beds	30.86	1.48	0.17	8.29	*
Rushes	1.00	0.05	0.17	32.15	*
Maize fields	2.25	0.09	0.17	29.11	*
Saltmarshes	9.00	0.72	0.16	2.90	NS
Reproduction					
Rice fields	0	–	–	–	–
Bocage	0	–	–	–	–
Reed beds	29.57	1.42	1.17	8.29	*
Rushes	0	–	–	–	–
Maize fields	0	–	–	–	–
Saltmarshes	0	–	–	–	–
NON-BREEDING PERIOD					
Presence					
Rice fields	19.80	3.27	0.09	636.54	*
Bocage	4.50	0.24	0.10	62.15	*
Reed beds	66.86	3.21	0.10	528.65	*
Rushes	11.00	0.60	0.10	17.49	*
Maize fields	3.38	0.14	0.10	79.25	*
Saltmarshes	29.57	2.37	0.09	208.66	*
Foraging					
Rice fields	14.40	2.38	0.15	82.03	*
Bocage	0	–	–	–	–
Reed beds	29.57	1.42	0.16	6.67	*
Rushes	1.00	0.05	0.16	34.04	*
Maize fields	0	–	–	–	–
Saltmarshes	5.14	0.41	0.16	13.56	*

TABLE 4

Summary of best models describing the relations between predictor variables and the presence and abundance of the marsh harrier, during the breeding and non-breeding periods, in Lower Vouga Lagoon, and the respective values of deviance, Akaike Information Criterion (AIC) and Akaike weights (wAIC) (for variable codes see table 2).

[Resumen de los mejores modelos que describen las relaciones entre variables predictoras y la presencia y abundancia del aguilucho lagunero, durante los periodos de reproducción y sin reproducción, en Baixo Vouga Lagunar, y valores respectivos de desviación, Criterio de Información de Akaike (AIC) y pesos de Akaike (wAIC) (para códigos de las variables véase la tabla 2).]

Models	Deviance	AIC	*w*AIC
PRESENCE			
Breeding period			
Intercept only	87.5	91.5	–
Spa_het + Maize + Reed + Sal + Apo + Road + QH	71.5	91.5	0.16
Spa_het + Maize + Reed + Sal + Road + QH	74.1	92.1	0.12
Spa_het + Reed + Sal + Road	80.2	92.2	0.11
Spa_het + Maize + Reed + Sal + Apo + Cattle + Road + QH	70.7	92.7	0.08
Spa_het + Sal + Road	82.8	92.8	0.08
Non-breeding period			
Intercept only	107.6	111.6	–
Maize + Reed + Apo + Cro + Mic + Atm + Road + Agr_Mac	55.8	75.8	0.26
Reed + Apo + Cro + Mic + Atm + Road + Agr_Mac	58.5	76.5	0.18
Reed + Apo + Cro + Road + Agr_Mac	62.8	76.8	0.15
Maize + Reed + Sal + Apo + Cro + Mic + Atm	55	77	0.14
Reed + Apo + Cro + Atm + Road + Agr_Mac	62	78	0.08
ABUNDANCE			
Breeding period			
Intercept only	248	252	–
Reed	244	250	0.14
Breeding + Reed + Cattle	240.1	250.1	0.14
Reed + Cattle	242.3	250.3	0.12
Breeding + Reed	242.4	250.4	0.12
Breeding + Reed + Vs + Cattle	238.5	250.5	0.11
Non-breeding period			
Intercept only	291.3	295.3	–
Forag + Reed + Mus + Cattle + QR	253.2	267.2	0.27
Forag + Reed + Mus + Rhum + Cattle + QR	251.5	267.5	0.23
Forag + Reed + Sal + Mus + Rhum + Cattle + QR	250.4	268.4	0.15
Forag + Reed + Cattle + QR	256.6	268.6	0.13
Forag + Reed + Sal + Mic + Mus + Rhum + Cattle + QR	249.5	269.5	0.08

TABLE 5

Standardised parameter estimates for the variables included in the best models describing the presence and abundance of the marsh harrier in Lower Vouga Lagoon during the breeding and non-breeding periods. Variable codes are available in table 2. Statistical significance: *** p < 0.001; ** p < 0.01; * p < 0.05; NS: non-significant.
*[Estimas de los parámetros estandarizados para las variables incluidas en los mejores modelos que describen la presencia y abundancia del aguilucho lagunero en Baixo Vouga Lagunar durante los periodos de reproducción y sin reproducción. Para códigos de las variables véase la tabla 2. Significación estadística: *** p < 0,001; ** p < 0,01; * p < 0,05; NS: no significativo.]*

Models/variables	Standardized estimate	Standard	Z Value	Pr(<|t|)
PRESENCE				
Breeding period				
Intercept only	1.26	0.35	3.66	***
Spa_het	0.00	0.00	17.73	***
Maize	0.03	0.01	3.84	***
Reed	0.04	0.01	6.53	***
Sal	0.02	0.01	3.80	***
Apo	-0.54	0.35	-1.55	NS
Road	0.00	0.00	-8.85	***
QH1	-8.37	0.86	-9.62	***
QH2	-9.31	1.75	-5.33	***
Non-breeding period				
Intercept only	7.92	2.87	2.76	**
Maize	0.01	0.00	1.61	NS
Reed	0.02	0.01	2.93	**
Apo	-0.26	0.09	-3.00	**
Cro	-3.10	0.70	-4.46	***
Mic	0.93	0.47	1.98	*
Atm	-0.08	0.03	-2.75	**
Road	0.00	0.00	3.66	***
Mac_Agr1	-5.01	0.97	-5.16	***
ABUNDANCE				
Breeding period				
Intercept only	0.49	0.16	3.08	**
Reed	0.00	0.00	2.18	*
Non-breeding period				
Intercept only	2.39	0.59	4.02	***
Forag	0.50	0.18	2.78	**
Reed	0.00	0.01	3.40	***
Mus	-0.10	0.06	-1.77	NS
Cattle	-0.87	0.31	-2.78	**
QR	-0.73	0.18	-4.11	***

and food availability (Martin, 1993; Němečková *et al*., 2008). Several studies report that marsh harriers use marshland vegetation in flooded areas, demonstrating a preference for reed beds (Thiollay, 1967; Schipper, 1978; Underhill-Day, 1985; Bavoux *et al*., 1990; Costa *et al*., 1993). Such a preference was also confirmed in the Lower Vouga Lagoon throughout the year, as reed beds were positively selected for foraging and during both the breeding and non-breeding periods and were also used for nesting. As stated by Cardador and Mañosa (2001) and Cardador *et al*. (2009), the patterns of foraging habitat selection of the marsh harriers may result from a compromise between the quality of the different patches and their distance from nesting sites. In our study, during the breeding season, both nesting and foraging occurred in reed beds. Thus we conclude that this habitat is very important for the species in this region. Factors contributing to the quality of reed bed habitat should include the seasonal flooding, which protects the nests against terrestrial predators; the density and height of the reeds, which protect from the wind; and food availability (small mammals (Alves, 2013) and birds (IDAD, 2008) in our study area). During the non-breeding period, rice fields, which actually offer higher food availability than reed beds (Alves, 2013), are correspondingly the preferred foraging habitat (table 3). In general, the habitat selection results demonstrated that in addition to reed beds and saltmarshes (natural wetlands), rice fields (during the non-breeding period) are the most important habitat for the occurrence of marsh harriers at the Lower Vouga Lagoon. Farmlands, with their typically high food availability for both prey and predators, short vegetation in the case of crops and irrigation may increase the accessibility of small mammals to marsh harriers (Cardador *et al*., 2011). However, maize fields were avoided by marsh harriers (table 3), despite being the most extensive habitat in the study area. Since crop height may influence foraging by raptors (Rodríguez *et al*., in press), at least during early stages of the crop development, one could expect that young maize could also provide good foraging conditions. However, maize fields offered low small mammal abundance (Alves, 2013), failing to provide food for harriers. These results agree with previous findings showing that although farming intensification is generally a major threat for open-habitat bird species (Benton *et al*., 2003), some intensive agricultural lands are still able to provide adequate foraging conditions for some species –such as the marsh harrier (Cardador *et al*., 2011)– at least outside the sensitive breeding season, and if food availability is not compromised (Ursúa *et al*., 2005). Moreover, field observations indicated that human disturbance, resulting from agricultural machinery, was greater and more continuous in maize than in rice fields, which may also have contributed to maize avoidance.

Presence and abundance of marsh harriers

We found that landscape and human disturbance variables were in general the most important variables influencing the presence and abundance of marsh harriers in the study area. Not only individual habitats, such as reed beds, saltmarshes and maize fields were important, but also the landscape heterogeneity presented a significant positive effect on the presence of the species during the breeding period. This suggests that the diversity of habitat opportunities may favour the occurrence of marsh harriers in the region, as demonstrated for other species (Rodríguez *et al*., in press) or taxonomical groups (Mendes *et al*., 2014). The consistent positive effect of reed beds throughout the year confirmed this natural habitat as a key element for the species in this region. Indeed, the importance of wetlands, and reed beds in particular, was already acknowledged in previous studies (e.g. Báldi, 1999; Cardador *et al*., 2011; Verma and Prakash, 2007).

Human disturbance variables, such as agricultural machinery, constructions, road occupancy and cattle, presented a general negative effect on marsh harriers. Previous studies (Cardador *et al*., 2011) found no relationship between human pressure and nesting-site occupancy, but our field observations showed clear disturbance and avoidance behaviour of birds when, for instance, farmers and machines were operating in the area. Roads had a significant negative influence on the presence of marsh harriers during the breeding period, but a positive effect during the non-breeding period (table 5), suggesting that this population may be sensitive to this type of human pressure only during reproduction. Roads can increase the presence of diurnal raptor species in a given area, thereby causing possible competition (Lambertucci *et al*., 2009). The presence of other diurnal raptor species, such as the black-shouldered kite *Elanus caeruleus*, common kestrel *Falco tinnunculus*, black kite *Milvus migrans* and Common Buzzard *Buteo buteo*, resting or hunting near or over roads, was in fact confirmed during fieldwork, but no territoriality or other competition behaviours towards marsh harriers was observed, so this hypothesis was not confirmed in this study.

The degree of disturbance caused by other human constructions, such as houses or warehouses, showed little or no relevance in the results, but we believe they must also be considered. In fact, the consequences of this type of disturbance are often difficult to detect and quantify, especially because they are not immediate. Yet, birds may be affected indirectly by them, for instance in terms of reproductive success (Fernández and Azkona, 1993).

Relevance for conservation

In general, the results lead to the conclusion that in the Lower Vouga Lagoon, the marsh harrier utilises anthropogenic landscapes such as rice fields to forage. However, human disturbance that influences the presence and abundance of the species may reach harmful levels if there is no proper management of the agricultural and wetland areas. The level of disturbance may determine the use or avoidance of certain habitats, despite the availability of favourable nesting or hunting conditions there. Due to high human pressure, individuals may be forced to seek less favourable locations, which may ultimately have a negative impact on reproductive success and thus lead to a population decrease.

The availability of food resources in agricultural areas may also be altered as a result of their intensive use (*e.g*. Burel *et al*., 2004), eventually leading to lower diversity and availability of prey for the marsh harrier. This condition emphasises the importance of preserving natural habitats, such as reed beds and saltmarshes, which are used by the species throughout the year. Since wetlands and water quality are strongly related, particularly within agricultural complexes (Verhoeven *et al*., 2006), this study also highlights the importance of protecting water resources in this humanised landscape. All factors that may influence water quality and availability, such as pollution and the use of chemicals associated with agricultural and industrial activities in the region, should be considered in a well-structured, integrated management plan. Since the marsh harrier is an umbrella species (e.g. Kaboli *et al*., 2006), its conservation may favour local biodiversity and nature conservation as a whole. However, the management of a humanised area may not be viable without the collaboration of the local residents. For instance, it would be relevant to make local landowners aware of the ecological dynamics of the Lower Vouga Lagoon, and the importance of preserving biodiversity. The sustainability of their agriculture, namely through the application of correct timings and practices (e.g. logging, ploughing, harvesting and reed cutting) that

are compatible with natural equilibria should urgently be promoted. We thus suggest the creation of a management plan aiming at sustainability and so favouring both economic activities and the conservation of natural values, while simultaneously promoting public awareness.

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1. Executive Summary

- This report provides a summary of the results of surveys undertaken from 2009 until 2016, evaluating them against the requirements of the Marine Licence and its stated objectives in order to determine whether these requirements have been discharged.
- A process, informed by the conclusions of the Environmental Statement (ES) and consultation with the Marine Management Organisation (MMO), Natural England and the Royal Society for the Protection of Birds (RSPB; organised through the Ornithological Review Panel [ORP]), determined the key species in relation to each monitoring objective. This report primarily presents analysis and conclusions with respect to the following effects and key species:
    - Barrier effects on divers;
    - Collision risk in relation to divers, gannet, large gulls, and small gulls; and
    - Displacement of divers and auks.
- Since 2009 APEM, on behalf of London Array Ltd., has conducted digital aerial surveys over the London Array Offshore Wind Farm (LAW) and associated control zones over the winter months November to February.
- The monitoring period began in November 2010 and concluded in February 2016. Zone 1 (which contained the LAW) and Zone 2 (surveyed as the 'reference' site), were consistently surveyed from 2010 until 2016.
- The period 2010 until 2016 was classified as the monitoring period with three phases: pre-construction (2010 to 2011), during construction (2011 to 2013), and post-construction (2013 to 2016). Annual monitoring reports were produced for the programme of surveys conducted each winter across this period.
- The survey months were selected to coincide with the key season for the occurrence of non-breeding red-throated divers, as agreed as part of the ORP process.
- The most abundant species group recorded was divers (peak density: 15.86 birds km^{-2} recorded for red-throated divers) and the second most abundant was auks (peak density: 5.58 birds km^{-2} recorded for guillemot / razorbills).
- Non-parametric analysis was undertaken for gannets, small gulls and large gulls to investigate if there was any significant difference in densities between the development phases for Zone 1 and Zone 2. A significant difference was observed for common gull (P=0.01) in Zone 1 and herring gull in Zone 2 (P=0.01).
- Analysis was not undertaken for species recorded sporadically (scaup, common scoter, unidentified seaduck species, fulmar, shearwater, grebes, waders and skuas) during the monitoring period (2010 to 2016) as too few individuals were recorded to provide a meaningful result.

1.1 Assessment of Barrier Effect Summary

- To assess barrier effects, the relative bearing of each diver to the closest turbine was estimated. Directions were subdivided in to four quadrants: towards, away, and two for flying parallel in relation to the nearest turbine based on the relative angle.
- Two approaches were used to test the relative direction of flying divers in relation to the nearest turbine of the LAW. The null hypothesis that divers were likely to fly in all directions was tested using first a chi-squared (χ^2) test and second a randomisation approach.
- The first analysis found no significant difference in the proportion of divers flying towards the nearest turbine of the LAW between two regions: the LAW to 4 km region and greater than 4 km region (P >0.05). However, the number of flying divers recorded was low, especially in the buffer regions nearest to the LAW.

- To further investigate, additional analysis of flight directions was undertaken based on data bootstrapping. The results of this analysis showed that in the 2 to 4 km buffer region, significantly fewer divers than expected by chance were recorded flying toward the wind farm ($P < 0.05$). For the remaining buffer regions, LAW to 2 km and greater than 4 km, no significant difference was found between the number of divers flying towards the OWF and in other directions
- No flying divers within the footprint and a trend for increasing densities of flying divers away from the LAW post-construction suggest avoidance that could be in part brought about by a possible barrier effect. The evidence that divers avoided flying towards the LAW when between 2 km and 4 km of it provides stronger evidence of a possible barrier effect created by the LAW.
- This analysis was specifically conducted in relation to individual flight direction relative to the nearest turbine, rather than in relation to areas of suitable habitat (e.g. within the Outer Thames Estuary Special Protection Area [OTE SPA]). The area covered by the OTE SPA consists of large areas of favourable habitat for divers and any effect on energy expenditure created by a barrier would be expected to be negligible as rather than flying around any obstruction the divers may be able to utilise other feeding or roosting areas in other directions.

1.2 Assessment of Collision Risk Summary

- The density of flying individuals within the LAW was calculated for each species for each survey month based on the number of individuals in flight and the area surveyed. The peak density estimates per development phase of each key species were identified and qualitatively compared between pre-construction and post-construction.
- The expected number of collisions in the LAW was extrapolated from the estimated number of collisions in the ES. Assuming that the flight behaviour of each species did not change between the pre- and post-construction phases, and that comparable survey data have been gathered for the two phases the change in the estimated number of collisions for each species in the LAW from that predicted in the ES would be directly proportional to (1) the change in the species' densities used for the ES and those recorded post-construction, and (2) the ratio of the total rotor swept zones used in the built wind farm to that estimated in the ES. If these assumptions are correct the following should be exact.
- Red-throated divers were not recorded in flight during the post-construction phase of the LAW and therefore the species is not expected to be at collision risk from the turbines. For gannet, lesser black-backed, herring gull, and great black-backed gull the estimated number of collisions is lower than that predicted in the ES primarily due to the reduced number of turbines installed at LAW compared to the number used in the ES predictions. The lower flying bird densities measured in the post-construction surveys compared to that included in the ES predictions also contributes to lower collisions estimates for all but great black-backed gull. These results do not support the need for further collision monitoring.

1.3 Assessment of Displacement Summary

- APEM Ltd was commissioned by Natural England in 2016 with permission from London Array Ltd to undertake spatial modelling of diver and auk density based on relevant environmental variables following the first year of post-construction surveys.

- The purpose of the modelling was to identify any potential displacement impacts and to set up a suitable model framework for any subsequent analysis of the data collected at the LAW.
- Using the same spatial modelling approach but including the final two years of post-construction surveys, the Centre for Research into Ecological and Environmental Modelling (CREEM) repeated the assessment of displacement using the MRSea package in the R statistical program. The modelling outputs were summarised and presented graphically according to distance from the LAW.
- The density profile of divers increased gradually throughout the buffer regions with a peak at 9 km pre-construction. Year-to-year fluctuations in diver numbers and distribution should be an important consideration when interpreting the results. The displacement distance estimated for divers was up to approximately 12 km based on changes in the modelled absolute densities of divers between pre- and post-construction survey periods..
- Complete displacement was not detected at any distance, but the area of influence on diver densities was greater than what was originally predicted in the ES. This was primarily due to the limited evidence for potential effect distances available in the literature at the time the ES was issued
- The modelling showed that an estimated 1,111 divers (approximately 6% of the population which is approximately twice the percentage assumed in the ES) have been displaced from an area within the OTE SPA centred around the LAW.
- A decreasing proportion of auks were estimated to be displaced up to approximately 5 km from the LAW but, as for divers, complete displacement was not detected at any distance.
- The modelling showed that an estimated 536 auks were displaced from an area within the OTE SPA centred around the LAW, with an area of influence of up to approximately 5 km.
- The areas surveyed, Zone 1 and Zone 2, formed an irregular buffer around the LAW. This means only the smaller buffer regions were completely surveyed and larger buffer distances would have been incompletely surveyed. It is not possible to state with any certainty what the displacement effects would be for the surrounding areas that were not surveyed.

BioOne COMPLETE

Importance of ecological compensation areas for small mammals in intensively farmed areas

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Importance of ecological compensation areas for small mammals in intensively farmed areas

Janine Aschwanden, Otto Holzgang & Lukas Jenni

Aschwanden, J., Holzgang, O. & Jenni, L. 2007: Importance of ecological compensation areas for small mammals in intensively farmed areas. - Wildl. Biol. 13: 150-158.

Small mammals make up an important link in the food chain as many predator species feed on them. There are indications that small mammal populations in Europe are declining due to the intensification of agriculture. According to national legislation, farmers in Switzerland have to cultivate at least 7% of their land as ecological compensation areas and, thus, some alternative habitats that are possibly beneficial for small mammals have been created. In this study, we estimated the diversity and density of small mammals on two types of conventional farmland field types (artificial grassland and autumn-sown wheat) and three types of ecological compensation areas (wild-flower strips, herbaceous strips and low-intensity meadows) by use of capture-recapture in March, May and July 2003. The common vole *Microtus arvalis* was the most abundant and predominant species in all habitat types except in herbaceous strips, which harboured the highest diversity with six species caught. In March the density of small mammals was generally very low, but significantly higher in wild-flower (mainly due to common vole) and herbaceous strips than in the other habitat types. In wild-flower and herbaceous strips, densities increased strongly from March to May and in July. On autumn-sown wheat fields, a strong increase occurred only from May to July and was caused by common vole. On artificial grassland and low-intensity meadows, densities of small mammals (mainly common vole) increased only marginally with low-intensity meadows supporting slightly higher densities. Thus, habitats that were not mown each year supported the highest densities of small mammals. This demonstrates that ecological compensation areas, such as wild-flower and herbaceous strips, make up an important refuge for small mammals. They probably also have positive effects on populations of many predator species that depend on small mammals, particularly if a mosaic with mown surfaces is created.

Key words: density of small mammals, ecological compensation area, intensive agriculture, Microtus arvalis, vole

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Small mammals constitute an important food resource for many avian and mammalian predator species. In Europe, the fossorial form of water vole *Arvicola terrestris* and many vole *Microtus* species are common and typical inhabitants of agricultural landscapes, yet many farming practices have negative short-term effects on these animals. Mowing and harvesting, for example, remove shelters and food (Tew & MacDonald 1993), and tillage destroys burrows. The intensification of agriculture in western Europe during the last decades may thus have resulted in a decline in vole populations and their predators. Indeed, there are indications that since the beginning of the 1980s the abundance of voles declined in agricultural landscapes of Germany and France (Kostrzewa & Kostrzewa 1993, Butet & Leroux 2001, Reichholf 2004). In these countries agricultural intensification is thought to have resulted in a decrease in the frequency and amplitude of the typical vole population cycles. A similar decline in vole populations may be assumed for agricultural areas in lowland Switzerland, although, to our knowledge, there is no study describing population dynamics of voles in this part of the country.

In order to counteract the loss of biodiversity in intensively farmed areas, farmers in Switzerland are bound by law to cultivate at least 7% of their land as 'ecological compensation areas', and they get subsidies for their ecological contribution (Harder 1998). Additional subsidies can be applied for if areas show a high ecological quality or are connected to each other (Oppermann & Gujer 2003). Legally approved ecological compensation areas, each with specific guidelines of cultivation and a subsidy, include for example: low-intensity meadows, litter meadows, hedgerows, wild-flower strips and traditional orchards.

Some ecological compensation areas like wild-flower strips and herbaceous strips are not mown or tilled each year. Potentially, such areas constitute a suitable habitat for small mammals typical of the farmland. On a farmland in northern Yorkshire UK, the total small mammal biomass in autumn, for example, was found to be three times higher on 6-m wide field margins than on conventional arable field edges (Shore et al. 2005). Untilled land strips serve as a refuge for small mammals during harvest of adjacent agricultural fields (Tew & MacDonald 1993, Baumann 1996, Tattersall et al. 1997). However, we are not aware of any study in which the density of small mammals in different types of ecological compensation areas and agriculturally used fields in their vicinity has been monitored over several months.

In our study, we examined whether ecological compensation areas support higher densities of small mammals than artificial grassland and autumn-sown wheat fields over the vegetation period from March to July. While most previous studies concentrated on one species or on one habitat type (Baumann 1996, Tattersall et al. 1997, Briner 2002, Shore et al. 2005) or applied different trapping designs for different habitat types, we used the same trapping design and a coherent capture-recapture analysis to estimate the densities of small mammals simultaneously on three different types of ecological compensation areas, on artificial grassland and autumn-sown wheat fields.

Material and methods

Study area and vegetation

We collected data in an intensively farmed plain (17 km^2) near Wauwil (47°10'N 8°02'E, 500 m a.s.l.) in central Switzerland during the summer of 2003. Since 1995, the ecological compensation areas in the whole region have been increased from 3.2 to 8% of the cultivated area (Graf 1999). In the study area, ecological compensation areas amounted to 4.3% of the agricultural acreage (low-intensity meadows 3.6%, wild-flower strips 0.4% and herbaceous strips 0.3%), while artificial grassland (49%), autumn-sown wheat (8.5%) and other cultures (38.2%) dominated the area.

The abundance of small mammals was studied on five different habitat types, three of which were ecological compensation areas (low-intensity meadows, wild-flower strips and herbaceous strips),

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and two were conventional farmland field types (artificial grassland and autumn-sown wheat). Wild-flower strips are arable fallows sown with seed mixtures of native plants. They were on average 15 m wide and 185 m long (i.e. covering 0.28 ha). Herbaceous strips consisted of different species of herbaceous plants: thistles *Cirsium* spp., common teasel *Dipsacus sylvestris*, St John's Wort *Hypericum perforatum*, common mallow *Malva sylvestris* and mulleins *Verbascum* spp. (5 × 320 m, i.e. covering on average 0.16 ha). They were bordered by hedgerows on one side and conventional field types on the other side (e.g. artificial grassland, maize fields or potato fields). Low-intensity meadows (64 × 100 m, i.e. covering on average 0.64 ha) in the study area are typical *Arrhenatheretum* (i.e. meadows dominated by tall oat grass). They are usually cut twice a year with the first cutting no sooner than 15 June. Application of liquid manure or other fertilizers is not allowed. Artificial grassland (0.88 ha on average) is dominated by white clover *Trifolium repens* and ryegrass *Lolium* sp. and form part of the ordinary crop rotation practice. It is cut at least five times between April and October and liquid manure is applied regularly. One artificial grassland was situated between a herbaceous strip, a maize field and two farm tracks. The second artificial grassland was bordered by farm tracks on three sides and by a potato field on the fourth side, and the third artificial grassland was situated between two farm tracks, another artificial grassland and a wetland reserve. Autumn-sown wheat fields which are harvested at the end of July covered an average area of 1.3 ha. They were machine harvested leaving little standing stubble. The wheat fields were mostly bordered by artificial grassland, potato fields and farm tracks.

For each of the five habitat types, three fields were chosen. During each trapping session (in March, May and July), the vegetation structure of each field was described on five randomly chosen squares (each of 1 m^2) as follows: a) vegetation height (in cm) was determined by measuring the 10th highest plant (to avoid measuring unusually high plants), b) vegetation density was determined by measuring the height (in cm) at which only 50% of a horizontal ruler, 1.7 cm wide and 100 cm long, was visible by watching vertically from above, c) percent cover of green vegetation, and d) dead plant material was estimated visually (at an accuracy of 5%).

Trapping of small mammals

The density of small mammals in each of the 15 fields (five habitat types with three replicates each) was determined using capture-recapture during three trapping sessions; one in March, one in May and one in July. In each field, we set 40 traps (Trip Trap with a nest box; we glued two metal washers to the outside of the trap door to ensure door shutting even during wet conditions). Because wild-flower and herbaceous strips were narrower than 10 m, we could not set traps in a square grid design. Therefore, the traps were placed along two straight lines 5 m apart. Along each line, 10 pairs of traps were set at 5-m intervals. Thus, the trapping rectangle covered an area of 45 × 5 m. In low-intensity meadow, artificial grassland and autumn-sown wheat, the borders of the trapping rectangle were at least 40 steps away from adjacent habitat types. In wild-flower and herbaceous strips, we were forced to set the trapping lines close to the edge of the strips.

We never set traps in freshly mown patches, and the vegetation height usually exceeded 20 cm, except for artificial grassland and low-intensity meadows in March when vegetation height was just below 20 cm. If there were indications of small mammal activity such as corridors or holes within a circle of a 50-cm radius around a trapping point, the traps were placed directly at such a sign. The openings of each trap pair were orientated into different directions. Each trapping session consisted of continuous trapping during 60 hours, starting in the evening (for three consecutive days and nights). Traps were checked every eight hours, starting at 23:00. Small mammals were lured with a piece of apple and a piece of mixed rolled oats, peanut butter and minced meat (Holzgang & Pfunder 2002), and a paper tissue was offered as nesting material. The animals were marked by locally cutting of the guard hair on the shoulder or on the back. Because the underhair is darker than the guard hair, the marking was visible as a dark spot in the fur. For every recapture round, the fur cutting was made at a different place (e.g. on the left shoulder, right lower back). By doing this, the time of the first and subsequent captures, respectively, were known for every recapture, and resulted in capture histories of eight time points for each trapping session.

Data analysis

The trapped individuals were assumed to live on a surface of 500 m^2 (50 × 10 m), which results from

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multiplying the length and width of the trapping rectangle after addition of a margin of 2.5 m on each side (Gurnell & Flowerdew 1994, Williams et al. 2002). The population size on each single field for each trapping session was estimated using closed capture-recapture models by use of the computer software CAPTURE (Otis et al. 1978). These models assume a closed population during the 60 hours of each trapping session, thus only capture probabilities needed to be calculated to estimate the number of individuals. We believe that the assumption of a closed population is valid, as it is quite improbable that there is much immigration or emigration within only three days. Separate analyses were performed for each field and trapping session. For 22 out of the 45 samples it was not possible to calculate population sizes by the use of CAPTURE due to too few recaptures. In these cases, the population sizes were estimated by use of the formula $N = C/(1 - (1 - P)^8)$ where C is the number of captures, P is the average of the capture probabilities (calculated using CAPTURE) of all samples of one trapping session and eight as exponent is the number of trap checks per trapping session. From the population estimates, we calculated the densities of small mammals (ha^{-1}) and the average density of the three fields sampled for each of the five habitat types with its standard error.

The common vole was the only species for which sufficient captures were obtained to estimate densities at a species-specific level. We estimated the proportion of common voles from the pooled overall density of all species of small mammals (see above) from the proportion of common voles caught and corrected for the species-specific capture probabilities as corrected proportion of common vole = (number of common vole/p (common vole))/(number of species 1/p (species 1)) + … + (number of species n/p (species n)), where p is the capture probability of each species pooled for May and July, respectively. This corrected proportion of common voles multiplied with the overall density of small mammals yielded the density of common voles and was calculated for each field.

Small mammal densities were analysed for their dependence on habitat type, month (trapping session), the interaction between habitat type and month and the four parameters of vegetation structure (height, density, cover of green vegetation and dead plant material) with an analysis of variance. Because the error variance did not show an equal distribution among the habitat types, an important criterion for the application of an analysis of variance was violated. Therefore, we used a weighted multifactorial analysis of variance (weighted least squares) according to Neter et al. (1990). The weight factor in such an analysis corresponds to the reciprocal value of the error variance.

Results

In total, we caught 349 different small mammal individuals at 738 occasions; thus an animal was on average captured twice. We caught the following six small mammal species: yellow-necked mouse *Apodemus flavicollis*, wood mouse *A. sylvaticus*, bank vole *Clethrionomys glareolus*, field vole *Mi-*

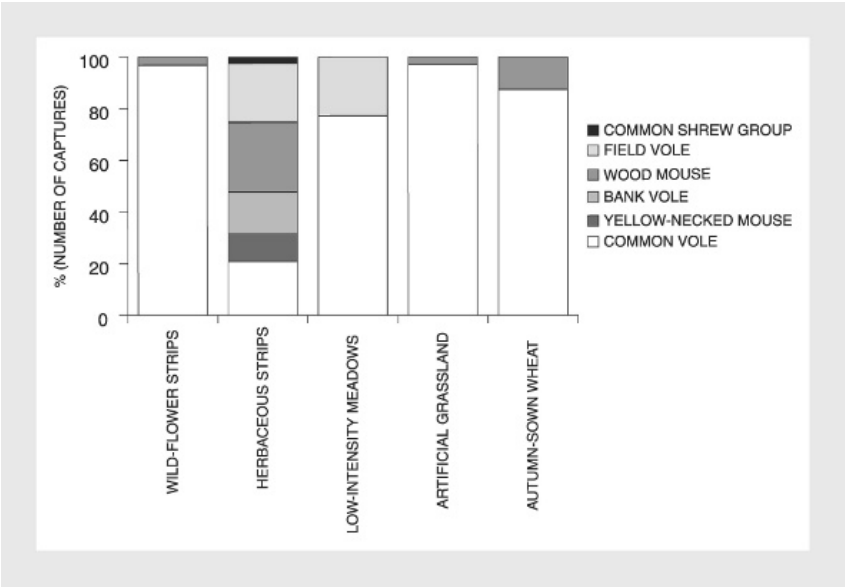


Figure 1. Species composition in the five habitat types, pooled over the three trapping sessions (N = 738 captures).

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Table 1. Weighted analysis of variance of small mammal densities with respect to vegetation parameters, month (trapping session), habitat type and the interaction of habitat type*month (adjusted $R^2 = 0.86$, df = 17, F = 15.5, $P < 0.001$). B is the parameter estimate and SE (B) is the standard error of parameter estimate.

	B	SE (B)	df	Mean square	F	P
Intercept	742.46	90.14	1	12.101	11.895	0.002
Vegetation parameters						
Cover of dead plant material	1.27	0.94	1	1.836	1.805	0.192
Cover of green vegetation	0.18	0.71	1	0.062	0.061	0.807
Vegetation density	1.34	1.35	1	0.994	0.977	0.333
Vegetation height	-0.31	0.52	1	0.368	0.362	0.553
Month			2	14.761	14.510	< 0.001
Habitat type			4	10.871	10.685	< 0.001
Habitat type*month			7	12.283	12.074	< 0.001

crotus agrestis, common vole *M. arvalis* and some individuals of the common shrew *Sorex araneus* complex. Herbaceous strips were the only habitat type in which all six species were captured (60% voles, 38% *Apodemus* spp. and 2% shrews; Fig. 1). In all other habitat types, > 75% of the catches were common voles together with one additional species, the field vole on low-intensity meadows, or the wood mouse on wild-flower strips, artificial grassland and autumn-sown wheat.

Vegetation parameters had no significant effect on small mammal density (Table 1) or common vole density (Table 2), even when entered into the model before month (trapping session) and habitat type. But both small mammal and common vole density differed significantly between months and habitat types (see Tables 1 and 2), and these two effects were not independent of each other. The significant interaction term habitat type*month shows that the increase in small-mammal density over the three months differed between habitat types.

In March, the density of small mammals was low (averaging 27.0 ha^{-1} ± 22.0 SE) over all habitat types (Fig. 2). Nevertheless, wild-flower and herbaceous strips had a higher density (57.6 ha^{-1} ± 22.3 and 88.4 ha^{-1} ± 68.7, respectively) than low-intensity meadows (0 ha^{-1}), artificial grassland (8.2 ha^{-1} ± 8.1) and autumn-sown wheat (8.2 ha^{-1} ± 8.1). In all habitat types, the common vole was the only species that we caught at this time of year, except for herbaceous strips where the small-mammal density was made up by wood mouse and yellow-necked mouse. In May, a substantial increase to 486.6 ha^{-1} ± 292.4 and 630.2 ha^{-1} ± 289.6 was observed on wild-flower and herbaceous strips, respectively, whereas in the artificial grassland, autumn-sown wheat and low-intensity meadows, the increase was clearly less prominent (to 61.8 ha^{-1} ± 67.4, 78.8 ha^{-1} ± 11.9 and to 113.3 ha^{-1} ± 50.0, respectively; see Fig. 2). The common vole was the most abundant species on wild-flower strips (92% common voles), low-intensity meadows (85%) and artificial grassland (100%), whereas it was only a by-catch on herbaceous strips (7.5%) and autumn-sown wheat (21%). Even larger differences in small mammal densities between habitat types were estimated for July. On low-intensity meadows and artificial grassland, the density had increased slightly from 113.3 ha^{-1} to 140.0 ha^{-1} (± 82.6) and from

Table 2. Weighted analysis of variance of common vole densities with respect to vegetation parameters, month (trapping session), habitat type and the interaction of habitat type*month (adjusted $R^2 = 0.91$, df = 18, F = 24.3, $P < 0.001$). B is the parameter estimate and SE (B) is the standard error of parameter estimate.

	B	SE (B)	df	Mean square	F	P
Intercept	656.91	55.30	1	142069	6.613	<0.001
Vegetation parameters						
Cover of dead plant material	0.86	0.67	1	35650	1.660	0.209
Cover of green vegetation	0.72	0.66	1	25554	1.190	0.286
Vegetation density	0.80	1.22	1	9235	0.430	0.518
Vegetation height	-0.09	0.20	1	4025	0.187	0.669
Month			2	615296	28.642	<0.001
Habitat type			4	293716	13.67	< 0.001
Habitat type*month			8	380734	17.723	< 0.001

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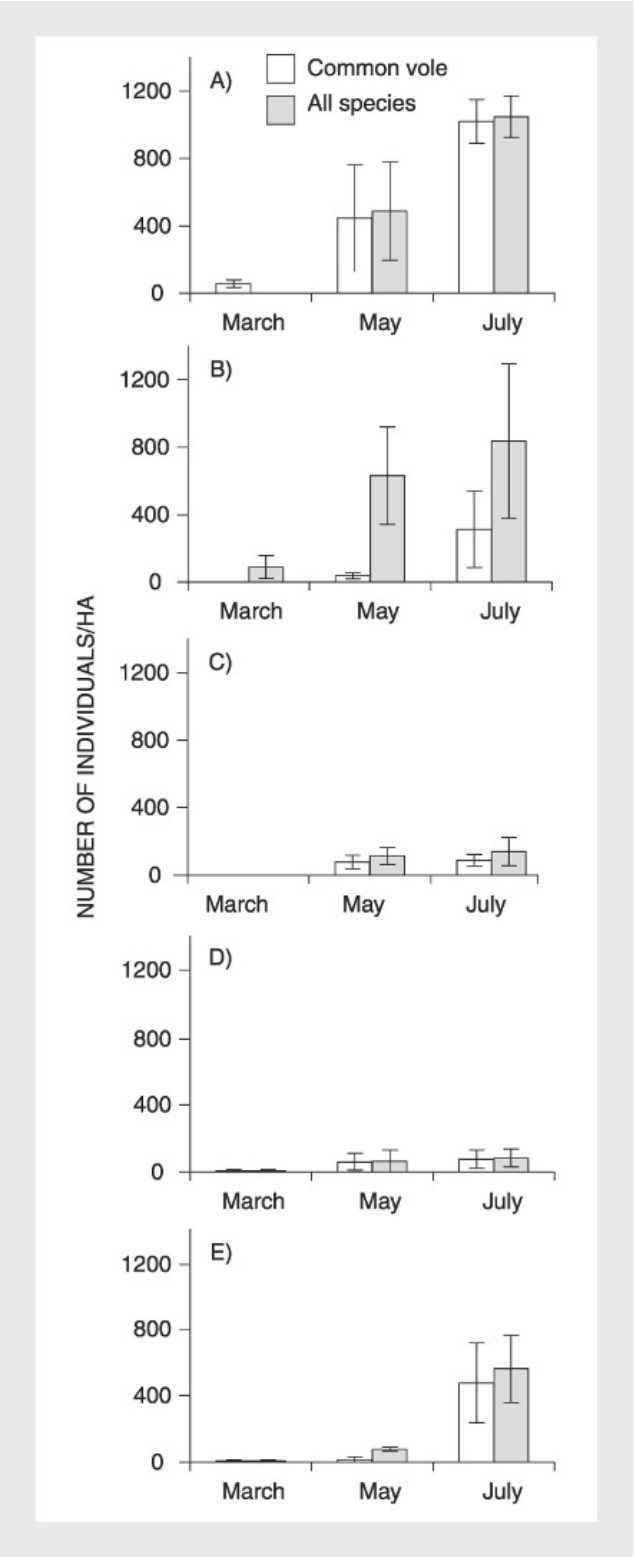


Figure 2. Mean densities of small mammal species (± SE) and mean densities of common voles (± SE), corrected for different capture probabilities, in the five habitat types (A) wild-flower strips, B) herbaceous strips, C) low-intensity meadows, D) artificial grassland and E) autumn-sown wheat) for March, May and July 2003. Capture probabilities: *M. arvalis* = 0.22, *M. agrestis* = 0.39, *A. sylvaticus* = 0.22, *A. flavicollis* = 0.25, *C. glareolus* = 0.19.

61.8 ha^{-1} to 84.0 ha^{-1}(± 54.0), respectively. In contrast, the densities of small mammals had increased further on wild-flower and herbaceous strips from May to July (from 486.6 ha^{-1} to 1046.6 ha^{-1} ± 123.5 and from 630.2 ha^{-1} to 836.4 ha^{-1} ± 457.6, respectively). During this period, an increase in density was also observed on autumn-sown wheat fields from 78.8 ha^{-1} to 561.3 ha^{-1} (± 201.7; see Fig. 2). In July, the proportion of common voles was > 80% of the total small mammal density in all habitat types, except in herbaceous strips where it amounted to 40%.

Discussion

Our study demonstrated large differences in species composition and seasonal abundance of small mammals between habitat types. The highest species diversity occurred in herbaceous strips with six species trapped. In all other habitat types, only two species were captured, with the common vole being the predominant species. Herbaceous strips are typical ecotones between hedges (including hedges along streams or channels) or woodland and agriculturally used land. This mosaic of different habitat types holds a diverse small-mammal community, and in the herbaceous strips we captured all the species typical for this habitat (Holzgang & Pfunder 2002). Clearly, bank vole, yellow-necked mouse and individuals of the common shrew complex, which prefer trees, bushes or dense undergrowth (Corbet & Ovenden 1982, Hausser 1995), moved from the adjacent hedge into the herbaceous strips where they were trapped. Also Baumann (1996) caught bank voles only in herbaceous strips adjacent to woodland. In wild-flower strips surrounded only by arable land, the common vole, typical of open land, was caught as well as the wood mouse, which moves also into open land with a good cover.

As expected, both the density of common voles and of all small mammals taken together generally increased over the study period from March to July. The common vole was the most abundant species on wild-flower strips, low-intensity meadows, artificial grassland and autumn-sown wheat, whereas it was only a bycatch on herbaceous strips. Although many tracks, particularly vole corridors, were observed in March, only a few individuals were caught. The estimated low density for March (27 ha^{-1}) corresponds to the findings of other studies, although the methods used differed. On fallows

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and orchards in West Germany, Boyce & Boyce (1988) estimated a vole density of 24 ha^{-1} for April 1982 (based on Minimum Number Alive). In western France, Butet & Leroux (2001) found vole densities between 0.8 and 87.6 ha^{-1} for April during 1986-1998 (pooled data based on the trap-night index for abandoned pastures, grazed and mown grassland and cereal crops). Gromadzki & Trojan (1971) estimated spring densities between 4.1 ha^{-1} and 33.8 ha^{-1} on cultivated fields and in refuge habitats in Poland in 1966 and 1967 (based on capture-marking-release).

The short vegetation might have offered only poor protection against predators (e.g. long-eared owl *Asio otus*, kestrel *Falco tinnunculus*, common buzzard *Buteo buteo*, red fox *Vulpes vulpes* and stoat *Mustela erminea*) and may have influenced the spatial behaviour of the small mammals resulting in reduced captures. Small mammals perceive tall vegetation as good protection (Tchabovsky et al. 2001). Under predation risk, which is high in short vegetation (Korpimäki et al. 1996, Longland & Price 1991, Jacob & Brown 2000, Sheffield et al. 2001), small mammals move faster, cover shorter distances (Lagos et al. 1995) and may reduce their home-ranges (Jacob & Hempel 2003). A reduced number of captures would influence the density of small mammal estimates negatively. For May and July, the effects of low vegetation cover can be excluded, as traps were only placed on fields with vegetation heights > 20 cm.

Compared to wild-flower and herbaceous strips, small mammal densities in autumn-sown wheat were very low in March and May, and increased only from May to July. The small mammals were probably exploiting this huge food resource shortly before harvest (i.e. at the end of July).

Densities of small mammals in artificial grassland remained at very low levels throughout the summer. Apparently, the regular cutting (first cut in April and afterwards about every five weeks until October) and subsequent application of liquid manure prevented the establishment of higher small-mammal densities. Populations of gray-tailed voles *Microtus canicaudus* on alfalfa fields are known to decline by about 50% after mowing (Edge et al. 1995).

Small-mammal and common-vole densities remained low for the whole study period in low-intensity meadows compared to wild-flower and herbaceous strips. Compared to artificial grassland, though, densities of small mammals and of common vole in May and July were twice as high. This may be the result of the lower frequency of mowing (only twice, first cut after 15th of June).

Many studies have shown that vole density is influenced by vegetation height, cover and litter (e.g. Ostfeld et al. 1985, Tattersall et al. 1997, Sheffield et al. 2001, Olson & Brewer 2003). Unexpectedly, we found no significant correlations between small mammal or common vole densities and parameters of vegetation structure in our study. Apparently, it was not vegetation structure, but habitat type that determined small-mammal densities in our study area. The habitat types differed in agricultural activities (mainly in the frequency of mowing and the application of liquid manure) and food availability (i.e. autumn-sown wheat). Therefore, it is likely that the density patterns of small mammals seen in our study area were mainly determined by the adverse effects of agricultural practices (i.e. mowing and liquid manure) and food availability.

Our study clearly demonstrated that ecological compensation areas not tilled or mown every year, such as wild-flower and herbaceous strips, support high densities of small mammals and can serve as a refuge in intensively farmed areas. Similarly, grassy field margins were found to harbour a higher small mammal biomass than conventional arable field edges in the UK (Shore et al. 2005). If connected to other semi-natural habitats (e.g. hedgerows), such ecological compensation areas may also increase the species diversity of small mammals. Regularly mowed ecological compensation areas (low-intensity meadows) support a lower diversity and density of small mammals.

The density of small mammals, in particular of the genera *Microtus* and *Arvicola*, directly determines the occurrence and abundance of many avian and mammalian predators. While some predators exploit small mammals opportunistically when their densities are high (e.g. red fox; Ferrari & Weber 1995, badger *Meles meles* and beech marten *Martes foina*; Niethammer & Krapp 2002), certain bird and mammal species are specialised predators that depend almost exclusively on them (e.g. stoat and weasel *Mustela nivalis*; Niethammer & Krapp 2002, long-eared owl; Mebs & Scherzinger 2000). Such specialised predators have a higher reproductive output or reproduce exclusively during periods of elevated small mammal densities (e.g. Wendland 1957, Niethammer & Krapp 2002). However, as shown in a study of the hunting behaviour of common kestrel and long-eared owl, it is not the

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absolute density of voles, but their accessibility, that determines where they hunt. Indeed, they mostly hunted on freshly mown grassland and also preferred freshly mown grassland bordering wild-flower and herbaceous strips (Aschwanden et al. 2005). Therefore, both refuge areas with high densities and a mosaic of freshly mown surfaces (preferably adjacent to high density areas) are important to support avian vole predators. Wild-flower and herbaceous strips are not only indirectly important for avian predators during summer, but also directly during winter as kestrels preferred wild-flower strips for hunting activities during the winter (Buner 1998).

In intensively farmed landscapes, ecological compensation areas, such as e.g. herbaceous and wild-flower strips, are of high value. As well as supporting highly productive and diverse small mammal communities, they support also a rich diversity and density of plants, insects and birds (Lys & Nentwig 1994, Jenny et al. 1998, Kleijn & Sutherland 2003, Walter et al. 2004) and thus are important to maintain biodiversity in agricultural landscapes.

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Biodiversity net gain. Good practice principles for development

A practical guide



CIEEM Chartered Institute of Ecology and Environmental Management

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Biodiversity net gain. Good practice principles for development

A practical guide

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Executive summary

Biodiversity is essential to sustain our society and economy. Enhancing biodiversity is integral to sustainable development, and BNG is an approach to embed and demonstrate biodiversity enhancement within development. It involves first avoiding and then minimising biodiversity loss as far as possible, and achieving measurable net gains that contribute towards local and strategic biodiversity priorities.

This guide offers practical advice to achieve BNG in the UK's land and freshwater environment. It applies to:

- building, infrastructure and extractive industries, as well as sites used by developers, such as construction compounds
- all sizes of development from a single home build to large-scale projects in rural and urban locations
- developments requiring consents such as planning permission, and those not requiring consent such as those under permitted development rights
- developments with limited or no adverse effects on biodiversity to projects in sensitive environments
- individual projects, a portfolio of projects or management of a land-holding or an estate
- new projects, those already underway and routine maintenance activities.

The guide is for:

- those involved with commissioning, planning, assessing, designing, constructing and operating development projects, whether in a policy, regulatory or advisory capacity
- those exploring possibilities to adopt BNG within their business
- local planning authorities (LPAs) seeking to incorporate BNG within local plans and planning decisions
- landowners and managers, including companies undertaking development on or managing an estate, as well as organisations supporting developers to achieve BNG.

This guide focuses on how to achieve BNG by following good practice, but it is not prescriptive in terms of approaches or tools. Rather, the advice is relevant to all development while its application will be proportionate to the scale of a project and its effects on biodiversity.

Developments involving Environmental Impact Assessments (EIAs), Ecological Impact Assessments (EcIAs) or Preliminary Ecological Assessments (PEAs) will already be undertaking many activities in this guide. Links with these assessments are identified throughout with reference to guidelines from IEMA (Ricketts *et al*, 2016) and CIEEM (2018) on EcIAs and PEAs. They are also summarised in **Technical notes T6 and T7**.

The guide does not contain technical advice on enhancing individual habitats or specific features, but should be read with such technical references.

BNG does not apply to statutory designated sites or irreplaceable habitats. This guide advises that impacts on statutory designated sites or irreplaceable habitats are avoided where possible. Any impacts should be addressed as fully as possible by adhering to the mitigation hierarchy, as well as legislative and policy requirements on a case-by-case basis, usually in collaboration with the planning decision maker and statutory nature conservation advisor.

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Case study 9.2
Engaging stakeholders on biodiversity net positive pilot for the Midland Mainline programme

The Midland Mainline Programme (MMP) involves permanent clearance of vegetation to enable the delivery of its electrification, building and civils projects (see Figure 9.2). Carillion, representing the MMP, recognised the need to address this loss by achieving biodiversity net positive (BNP). It ran the first engagement workshop with key stakeholders including Natural England, RSPB, the Environment Agency, Wildlife Trusts and representatives from the farming community (see Figure 9.3). The workshop included discussions on critical aspects of BNP, including local wildlife priorities that MMP could contribute towards. By holding the workshop at an early stage of MMP, the programme benefitted from being able to target its BNP activities and from establishing good working relations with key stakeholders.

Fore more details see Case study 9 in C776b.



Figure 9.2 Vegetation clearance

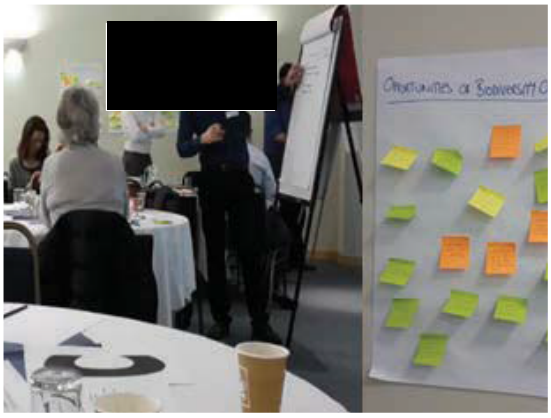


Figure 9.3 Stakeholder engagement workshop

9.5 ASSESS FEASIBILITY

An ecological scoping assessment, stakeholder consultations and PEA can inform an assessment of whether delivering BNG for the project is feasible. This can be an iterative assessment that is updated during the design stage when more information about the project becomes available. It can include the following:

- **Identifying statutory designated sites or irreplaceable habitats.** Projects negatively affecting statutory designated sites or irreplaceable habitats cannot, as a project, achieve BNG. **Technical note T3** gives advice on irreplaceable habitats.
- **Assessing risks**. **Chapter 11** lists risks to achieving BNG. Considering these at an early stage can help to plan (and demonstrate) application of the mitigation hierarchy. This is important because committing to BNG does not negate the need to first avoid and then minimise a project's loss of biodiversity. It also enables planning for a more efficient and less risky design stage.

  Highlighting risks to achieving BNG can inform a feasibility assessment of the project itself, especially when different options are being considered. For example, this could be identifying locations where a project will affect highly biodiverse features that, to address the effects, will add substantially more time and cost to the project.
- **Assessing wider benefits**. BNG can generate benefits for society, the environment and the economy, in addition to the goal of BNG. A high-level assessment of these wider benefits might be possible at an early stage, and can position BNG as part of the project's legacy. It can also identify additional benefits for the commissioning agency or landowner, for example making an estate more resilient to climate change.
- **Estimating biodiversity losses and gains**. The method for measuring a project's biodiversity losses and gains should be specified, with justification. For example, some commissioning agencies and LPAs specify a method, or if no method is specified, projects might use a commonly applied metric such as Defra's biodiversity metric (see **Technical note T9** for advice).

  Chapter 11 gives advice on measuring a project's biodiversity losses and gains. At an early stage, findings of an ecological scoping assessment or PEA might enable a calculation of estimated biodiversity losses to:

  - ☐ focus attention on measures to avoid and then minimise biodiversity loss

- highlight possibilities for on-site gains, eg estimating gains from enhancing habitats within the site footprint to compare with the time and cost implications of removing them, and the missed opportunity for achieving the gains
- produce a ballpark figure of the area and types of habitat to create or enhance to achieve BNG, which will in turn support planning for the budgets, resources and timescales (eg budgeting to create 10 ha of species-rich meadow, and maintaining this over a 25-year period).

When estimating biodiversity loss (see **Box 9.3**), milestones for updating this calculation should be set within the project programme. This aligns with the CIEEM guidelines, which describe how to analyse information gaps during a scoping assessment in order to plan and prioritise information gathering for the full EcIA.

- **Assessing resource, budget and programme requirements**

This assessment can help inform budgets for designing, implementing, maintaining and monitoring BNG. It can include:

- **data collection**, especially any new data required for a biodiversity metric during both the design and construction stages. Data on habitat condition for Defra's biodiversity metric is not typically collected for EcIA surveys, but can be collected during EcIA or other similar surveys
- **specific expertise**, eg surveyors with the right botanical skills, ecologists with experience in designing BNG, local knowledge of the ecological suitability of sites marked for habitat creation, and expertise for implementing the BNG design
- **timescales**, eg implementing a BNG design and its long-term maintenance
- **seasonal constraints**, eg affecting when data should be collected and when BNG activities can be implemented (although these could be similar to seasonal constraints affecting protected species surveys and mitigation measures, and landscaping schemes)
- **cost estimates to implement the BNG design**, eg habitat planting or enhancement (see **Section 9.5**, point 3), and management plans.

Box 9.3
Examples of options for delivering biodiversity net gain using Defra's biodiversity metric

In the worst case, the project could result in permanent loss of 38 biodiversity units of a scrub and grassland mosaic. For budgeting purposes only, this indicative calculation at the project's feasibility stage shows that the project could achieve BNG by Options A or B:

Option	Hectares	Habitat	Work required	Biodiversity net gain	Assumptions
A	6.5	Amenity grassland	Converting to scrub/grassland mosaic in moderate condition	49 biodiversity units	No spatial risk, a low difficulty risk and 10 years to reach target condition
B	7	Scrub/ grassland mosaic	Enhancing from poor to moderate condition	46 biodiversity units	No spatial risk, a low difficulty risk and 5 years to reach target condition

This indicative calculation is only to illustrate possible options for BNG. It is intended to support budgeting and programming, as part of the project's feasibility assessment. It will be updated during the design and EcIA stage, when the mitigation hierarchy will be properly applied.

9.6 SET A GOAL

Where achieving BNG is feasible, a statement on the project's BNG goal should be included in the report.

The goal can be a commitment or an aspiration if uncertainty exists, for example on resources. It can be high-level, reflecting any targets set by the commissioning agency and LPA, as well as local biodiversity priorities. It can also outline the intention to follow the good practice principles (see **Box 9.4**).

A goal shows the project team what is to be achieved, helping to get BNG budgeted and in the project programme.

Chapter 6 advises on setting BNG goals and targets that contribute towards biodiversity strategies.

(based on specific, individual features building on the project's ecological impact assessment, while refining any previous goals set for BNG, eg during a project's feasibility stage).

2 How these net gains contribute towards local and strategic priorities for biodiversity (building on the assessment of biodiversity priorities and including any targets set by the commissioning agency and LPAs).

See **Box 11.1**.

Box 11.1
Example of setting high-level biodiversity net gain outcomes during the initial design

Net gains in specific features affected by the project

The project will create new wetland habitat that provides more and better shelter and foraging resources for the wetland birds affected by the project after following the mitigation hierarchy. The project will achieve a minimum of 20% increase in biodiversity (measured in biodiversity units) through its wetland habitat creation, compared against the project's baseline.

Contributing towards biodiversity priorities

The project's wetland habitat creation will contribute towards recovery plans for wetland birds of conservation concern. The project will also create patches of marshy grassland to connect to other wetlands within the local biodiversity opportunity area, thereby increasing ecological connectivity within the locality.

For projects involving EIAs, the EIA co-ordinator can engage the engineering design team to integrate BNG outcomes throughout the design, as part of an iterative process whereby design is informed by impact assessment.

Advice for local planning authorities

Before planning proposals are submitted, working with developers on their design for BNG can secure a high quality planning application, for example by highlighting local priorities that their BNG design can contribute towards.

11.4 DESIGN FOR NET GAINS IN BIODIVERSITY

Designs for BNG are based on individual features in both qualitative and quantitative terms. Key considerations include the following:

- Should the net gains be in the same or a different type of habitat?
- Where should BNG activities be located?
- Should BNG enhance existing or create new habitat?
- How can BNG create more, bigger, better and joined-up areas for wildlife?
- How can BNG optimise wider benefits?
- Are the BNGs more than what would have happened anyway?

The following sections describe these considerations. While each is listed individually, in practice they will be complementary and influenced by practicalities such as the availability of suitable sites.

In all instances, designs should be to generate the best long-term outcomes for biodiversity taking into account all considerations in this guide.

The good practice principles call for decisions that are based on evidence, application of the mitigation hierarchy and that involve stakeholders. CIEEM (2018) says that *"evidence should be provided to support the likelihood of delivering the predicted [biodiversity] benefit"*. Accordingly, designs that adhere to the principles and these guidelines should contain justification of decisions made, with evidence where possible.

Advice for local planning authorities

The advice in this section should be used to:

- review a project's BNG design to ensure that it delivers optimal benefits as far as possible
- establish a plan-led approach to BNG that incorporates all key considerations
- produce guidance for developers on BNG that is specific to the administration area.

11.4.1 Should the net gains be in the same or a different type of habitat?

When compensating for biodiversity loss, CIEEM (2018) refers to compensating for the same type of features as those affected, and seeking to achieve at least equivalent levels of ecological functionality.

When designing BNG, good practice is achieving net gains in features of the same or higher biodiversity value as those affected by the development. It is also to achieve at least equivalent or better levels of ecological functionality and, overall, to improve the extent or condition of biodiversity affected by a project (or the biodiversity within or surrounding a project footprint if no negative impacts are incurred). See **Table 11.2**.

Table 11.2 Design aim/description

Design consideration	Biodiversity net gains
Habitats of high biodiversity value	In the same type of high-value habitat. Net gains in habitats of high biodiversity value should be in the same type of habitat. Habitats of high biodiversity value include those identified as being of principal importance for biodiversity in accordance with the Natural Environment and Rural Communities Act (2006). These are often referred to as 'priority habitats' and can be identified using data from the MAGIC website or JNCC (2011). Arable field margins specifically managed for wildlife should be considered as priority habitats. For those using Defra's biodiversity metric, habitats of 'high' distinctiveness should be considered as priority habitats.
Ecologically equivalent or better	In the same habitat as that affected by development, or a habitat of a higher biodiversity value that supports the same species affected. Generate the same or better levels of ecological functionality. **Box 11.2** describes the concept of 'ecological equivalence'. Net gains should be in the same habitat as that affected by development, or a habitat of a higher biodiversity value that supports the same species affected, for example where a project results in the loss of semi-improved grassland and achieves net gains by creating unimproved grassland. For those using Defra's biodiversity metric, net gains for losses of medium distinctiveness habitats should be achieved in habitats of medium or high distinctiveness. Whereas net gains for losses of low distinctiveness habitats should be achieved in habitats of medium or high distinctiveness. A BNG design should generate the same or better levels of ecological functionality. Ecological functionality is often not captured within biodiversity metrics but should be captured within qualitative assessments and can be essential to achieve BNG (see **Box 11.3**).
No trading down	Improve the extent or condition of biodiversity and do not result in lost or damaged features being replaced by features of lower biodiversity value A BNG design should improve the extent or condition of biodiversity affected by a project. It should not result in lost or damaged features being replaced by features of lower biodiversity value. This is regardless of whether a metric shows an increased amount of biodiversity after a project compared with the baseline. A BNG design should demonstrate the improvement in the extent or condition of specific features in qualitative terms. This should be accompanied by a quantitative assessment that shows that the numerical outcome for individual features is a net gain compared with the baseline.

Box 11.2
The principle of ecological equivalence within biodiversity offsetting

Equivalence is the principle that biodiversity offsets should provide habitat, functions, values and other attributes that are similar in type and proportionate to those affected by the project. These are referred to as 'in-kind' offsets where the offset is the same kind of biodiversity components in a similar ecosystem to that affected by the project.

There may be instances where 'out-of-kind' offsets are appropriate, ie offsets are in a different habitat from the habitat affected, but one of a higher biodiversity value. Out-of-kind offsets should demonstrably provide a greater contribution to landscape-level conservation goals (eg should better address the past disproportionate losses to specific habitat types). Maron *et al*, 2016.

Table 11.3 Examples of creating biodiversity net gain in either the same or a different type of habitat after applying the mitigation hierarchy

Example	Outcome
Creating gains in a different type of habitat so that greater benefits are achieved	A development results in losses of species-poor, semi-improved grassland that is locally common. The grassland did not support notable wildlife populations nor did it provide any key local ecological function. There is no option to achieve BNG within the development footprint.
Using Defra's biodiversity metric to achieve gains in a different type of habitat that has a higher distinctiveness than the habitat lost	The project borders a wetland reserve that is in decline, so addressing this decline and enhancing the wetland are priorities in the BAP. In agreement with the statutory conservation advisor and LPA, the project achieves BNG by enhancing the wetland and supporting ongoing management of the reserve. This contributes towards wetland targets in the BAP, and improves access to higher quality nature for neighbouring communities. Using Defra's biodiversity metric, the species-poor, semi-improved grassland scored low distinctiveness. The aquatic habitats within the wetland scored high distinctiveness.
Using Defra's biodiversity metric to achieve gains in the same type of habitat with a higher distinctiveness band	A project results in the loss of semi-improved acid grassland with a low cover of wildflowers. This habitat scored medium distinctiveness using Defra's biodiversity metric. The project achieves BNG by establishing new areas of species rich grassland with a high cover of wildflowers within the development footprint. This habitat scores high distinctiveness.
Creating gains in the same ecological function as that lost, through a different habitat type with higher biodiversity value	A transport project in an urban environment is seeking to achieve BNG. It required the removal of buddleia, which scored low distinctiveness using Defra's biodiversity metric but provided vital resources for local butterfly populations. BNG is achieved by planting linear swards of native, nectar-rich plants along the site boundaries that increase habitat for the butterflies and contribute towards targets on increasing resources for butterflies in the BAP.
Creating gains in the same ecological function as that lost from a project through the same type of habitat but of higher conservation value	A development is converting derelict land covered by species-poor grassland into housing. Common lizards occupy the site and their numbers are significant locally. Increasing cover and quality of acid grassland is a target in the BAP – for the habitat itself and to boost local populations of common lizard. The project enhances the condition of acid grassland nearby the development and creates new areas of acid grassland in locations identified in the BAP. The project commits funding to implement a 30 year management plan for all areas. This plan includes monitoring for the project to claim BNG when the newly created habitats have reached their target condition and when lizards have colonised the areas.
Designing BNG based on habitats and ecological function	A project is seeking net gains for the unavoidable loss of a hedgerow. The ecologist assesses the ecological connectivity of the hedgerow and uses the assessment to create new hedgerows that increase ecological connectivity within the landscape.

11.4.2 Where should biodiversity net gain be located?

CIEEM (2018) directs compensation of biodiversity loss to be *"as close as possible to the location where effects have occurred and benefit the same habitats and species as those affected."*

When designing BNG, keeping the net gains local to the development while contributing to local and strategic biodiversity priorities can be important. Key considerations can be:

- home ranges and/or ecological networks of species affected by the project
- areas identified of local and strategic priority for biodiversity, eg within local biodiversity plans and GI strategies.

Table 11.4 helps to decide where to locate BNG activities to generate the best outcomes for biodiversity. When conflicts exist between delivering local net gains and supporting wider priorities, the aim should be to generate the best outcomes for biodiversity that are commensurate with and proportionate to a project's effect on biodiversity.

Technical note T5
Aligning biodiversity net gain with the development consent order process

Summary

This technical note relates to the development consent order (DCO) process in place in England and Wales. The target audience for this technical note is those involved in nationally significant infrastructure projects (NSIPs) in England and Wales, who are following the DCO process. Similar large-scale projects in Scotland and Northern Ireland are dealt with by the relevant LPA.

T5.1 WHAT IS THE DCO PROCESS?

Development proposals requiring planning permission are normally dealt with at the local level by the relevant LPA. Recognising the complexity of large-scale infrastructure projects, the Planning Act 2008 put a new decision-making process in place for NSIPs in England and Wales.

A DCO for NSIPs is a legally binding permission for the project given by the relevant Secretary of State. A transport project, for example, would be determined by the Secretary of State for transport. Once given, a DCO defines the scope and extent of the scheme to be constructed, contains all powers necessary to construct the scheme and sets out requirements that have to be complied with.

The DCO process is led by a planning inspectorate and includes pre-application requirements, examination of the proposal by the planning inspectorate and a recommendation made to the Secretary of State for their final determination. As with locally determined development projects, there is a set period and process to challenge any decision made.

The types of infrastructure projects that are determined by the DCO process include waste and water infrastructure, pipelines, electricity infrastructure and large-scale road and rail projects.

T5.2 CAN BIODIVERSITY NET GAIN BE APPLIED TO NSIPS?

Yes. NSIPs typically affect large geographical areas. While potential effects on biodiversity must be addressed, NSIPs are unique opportunities for biodiversity gains over a large area. Projects of national significance should underpin and support the delivery of UK and devolved government biodiversity commitments and should focus on landscape-scale gains and a contribution to national biodiversity targets.

BNG for NSIPs should be discussed with those involved in early consultations, such as LPAs, statutory consultees and local wildlife organisations. LPAs with policies on BNG should highlight these to the developer and the planning inspectorate.

The business case for BNG (see **Chapter 5**) is amplified for NSIPs. The length of the project life cycle and large-scale nature of NSIP can result in a complex project with a considerable number of developers, contractors, consultants and landowners involved over time. By establishing BNG targets as an objective of the project as a whole, each party can promote their own contribution to the overall achievement of BNG.

T5.3 WHICH STAGE OF THE DCO PROCESS IS MOST RELEVANT FOR BIODIVERSITY NET GAIN?

The pre-application stage. This is the first stage in the DCO process. Unlike local planning applications, the applicant must undertake all the necessary consultation before submitting the DCO application. The application must demonstrate that consultee responses have been given appropriate regard. Undertaking and responding to consultations during the pre-application stage is the ideal opportunity to identify, promote and build in BNG. The six stages of a DCO application are shown in **Figure T5.1**.

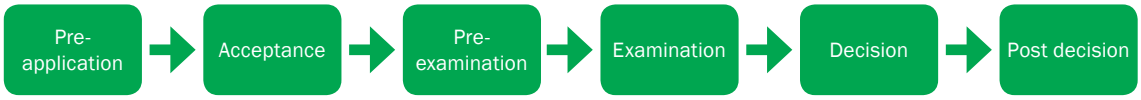


Figure T5.1 The six stages of a DCO application

T5.4 MAXIMISING OPPORTUNITIES AND BENEFITS AT THE PRE-APPLICATION STAGE

The DCO applicant must demonstrate that a comprehensive programme of consultation has been undertaken. The advice on stakeholder engagement within this guide will help ensure that the consultations are effective, in terms of exploring aspirations and options for BNG (see **Chapter 7 and Technical note T4**). Promoting the project as a BNG project and making a public commitment may help generate initial interest. Then including stakeholders in the development of BNG options can generate support for the project. A dedicated register of commitments will help to make sure that everyone understands what is being proposed for BNG.

At the pre-application stage, a DCO applicant should advertise the consultation and engage with stakeholders. BNG can be introduced as part of the project proposal at consultation events, including those focused on the local communities that may be interested in the proposal. The applicant may wish to consider using the guidance on corporate biodiversity targets and commitments within **Chapter 5**.

The applicant should consult with all relevant LPAs, statutory consultees and any people affected by the project. This gives an opportunity to engage with those who will have a specific interest in BNG. This could include wildlife organisations able to assist with locally relevant information and ideas.

During the pre-application stage, the applicant should build in the good practice for BNG as set out in this guide. **Chapter 8** on undertaking quality assurance, **Chapter 9** on feasibility and scoping, and **Chapter 11** on design will be particularly relevant.

Once a DCO application is in the pre-examination and examination phases, the benefits of early engagement will be realised, as these are the stages where consultees can make representations to the planning inspectorate. The DCO applicant can demonstrate that local biodiversity strategies have been taken into account, the mitigation hierarchy has been followed and BNG targets committed to within budgets and the project programme.

NSIPs are planned for over long time periods and there is an opportunity to build in BNG early so that time-lag between losses and gains is prevented where possible.

T5.5 HOW DOES THE LENGTH OF THE PROJECT LIFE CYCLE FOR NSIPS AFFECT BIODIVERSITY NET GAIN?

NSIPs are often in each stage of the project life cycle for several years. The construction phase alone can be extensive. The following good practice will help to keep the BNG target on track:

- Provide a dedicated resource for delivering BNG. This role should include co-ordinating, quality assurance checking and communicating (internally and externally), with full handover where there are staff changes.
- Start BNG activities early to avoid a time-lag between vegetation clearance for the project and biodiversity gains (and to minimise the penalties from time-lags when measuring the biodiversity gains).
- Apply a BNG quality assurance protocol at each life cycle stage.
- Ensure that all involved with each life cycle stage are fully aware of the BNG target and understand their roles and responsibilities.
- Set a clear BNG goal at feasibility and scoping. Given the length of time taken to establish a project design and submit a DCO proposal to the planning inspectorate, be flexible in the way that the goal is achieved. Allow the goal to benefit from any changing environmental conditions and new unforeseen opportunities that may arise during the pre-examination stages, such as creating habitat linkages as part of a newly published habitat restoration plan in the local area.
- Establish a BNG technical group to oversee the project life cycle. Include local biodiversity organisations to benefit from local expertise as well as new ideas and opportunities over time that may be relevant to any adaptive management (see **Chapter 13**).
- Given the lengthy timeframes, maximise the opportunities to create a legacy by, for example, long-term collaborations with local landowners and local wildlife organisations.

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Do Bat Gantries and Underpasses Help Bats Cross Roads Safely?

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Abstract

Major roads can reduce bat abundance and diversity over considerable distances. To mitigate against these effects and comply with environmental law, many European countries install bridges, gantries or underpasses to make roads permeable and safer to cross. However, through lack of appropriate monitoring, there is little evidence to support their effectiveness. Three underpasses and four bat gantries were investigated in northern England. Echolocation call recordings and observations were used to determine the number of bats using underpasses in preference to crossing the road above, and the height at which bats crossed. At gantries, proximity to the gantry and height of crossing bats were measured. Data were compared to those from adjacent, severed commuting routes that had no crossing structure. At one underpass 96% of bats flew through it in preference to crossing the road. This underpass was located on a pre-construction commuting route that allowed bats to pass without changing flight height or direction. At two underpasses attempts to divert bats from their original commuting routes were unsuccessful and bats crossed the road at the height of passing vehicles. Underpasses have the potential to allow bats to cross roads safely if built on pre-construction commuting routes. Bat gantries were ineffective and used by a very small proportion of bats, even up to nine years after construction. Most bats near gantries crossed roads along severed, pre-construction commuting routes at heights that put them in the path of vehicles. Crossing height was strongly correlated with verge height, suggesting that elevated verges may have some value in mitigation, but increased flight height may be at the cost of reduced permeability. Green bridges should be explored as an alternative form of mitigation. Robust monitoring is essential to assess objectively the case for mitigation and to ensure effective mitigation.

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Introduction

Recent research shows that major roads can have a major negative impact on bats. Bat density and diversity have been found to decline in proximity to a major road, with the scale of the impact indicating a barrier effect [1]. Studies of *Myotis bechsteinii* also provide evidence for a barrier effect with contracted foraging areas and reduced reproductive success of bats roosting close to roads [2]. Road avoidance behaviour has been observed in commuting bats [3], and street lighting [4] and traffic noise [5] reduce foraging activity and success. Numerous studies show that bats are killed on roads by collision with vehicles e.g. [6,7,8,9]. Roadkill is hard to quantify due to the difficulty of finding carcasses [10], but low reproductive rates make bats particularly vulnerable to elevated adult mortality e.g. [11].

Roads are detrimental to a wide range of animals, reviewed by [12,13]. Crossing structures, both under and over roads, have been built in an attempt to maintain connectivity across the landscape, and numerous studies report use of these structures by mammals and reptiles e.g. [14,15,16,17]. However, the use of a mitigation feature, the widely accepted criterion for success, does not make it effective. To be effective it must play a significant role in maintaining local population sizes. Evidence for a small, unspecified proportion of individuals using a structure to cross a road is not evidence for effective mitigation if a greater proportion crosses the road unsafely, is forced to reduce home range size or has to make longer journeys to find an alternative route. Reviews of mitigation techniques for a wide range of animals report that studies assessing use are typically qualitative [18,19]. In one such review, only two out of 123 studies were able to conclude a positive effect of mitigation at the population level [20].

European bats are protected by EU and national laws, see [1]. Legally required mitigation measures on road developments usually take the form of crossing structures to maintain linear elements in the landscape that bats rely on for commuting. The use of wire bat bridges or 'gantries' is becoming increasingly common: at least eight have been built in the UK in the last five years and six more are planned for the A11 in Norfolk [21,22]. However, there are no published data regarding the effectiveness of these structures. Recent reviews of case studies of bat mitigation in the UK found that most reports were at best qualitative and inconclusive [22,23]. Green bridges, underpasses and culverts have been installed across Europe with potential use as a wildlife passage frequently being an unintended or secondary function. Most of the studies reporting their use by bats are unsuited to quantitative analysis, or fail to address the important distinction between use and effectiveness e.g. [24,25,26]. Seven bat species were caught flying through motorway underpasses in Germany, but when activity levels were compared with sites in the

surrounding forest, only *Barbastella barbastellus* and *M. nattereri* were caught significantly more often in the underpasses, suggesting their effectiveness as crossing structures may be species-specific [2]. The use of underpasses by at least six bat species was also reported in Ireland, with the tendency to fly through the underpasses rather than over the road being related to the degree of clutter-adaptation of a species [27].

Our aim was to examine whether road crossing structures built for bats (or considered suitable for bats) are not only used but, moreover, are effective in guiding a significant proportion of bats safely over or under roads. The ideal study would determine the effectiveness of the structures in maintaining local bat population sizes, but this requires pre-construction data, which do not exist. We therefore studied their effectiveness in protecting crossing bats by reducing the risk of collision mortality. We studied underpasses, the most common wildlife crossing structure in Europe and North America, and wire bat gantries, bridge-like structures designed to guide echolocating bats over the road (see method for a detailed description). These are currently favoured in the UK and are also being built in other parts of Europe. They are sometimes referred to as bat bridges, but we have avoided this term to avoid confusion with other structures, such as green bridges.

Materials and Methods

Study Sites

All four study sites were located in northern England: three roads in Cumbria (A590 at 54°14′N, 2°55′W; A595 at 54°35.2′N, 3°33′W; A66 at 54° 38.3 N, 3°31.4′W), and one in Northumberland (A69 at 54°58′N, 2°15′W). All sites were located in rural lowland used primarily for agricultural grazing, with linear elements such as hedgerows, dry stone walls and tree lines providing connectivity for bats [28,29]. The importance of all sites as bat foraging and commuting routes was established during pre-construction environmental impact assessment, see Appendix A in [22], but methodological differences and inadequate data in these assessments prevented comparison with this study. All roads were built to bypass nearby settlements with traffic volumes of 12,000–17,000 vehicles per day [30,31]. All bat gantries were of similar design: two wooden or metal pylons erected at either side of the road with 2 or 3 pairs of wires spanning the road between them (approximately 20 m on two lane roads and 30 m on four lane roads), with plastic spheres at intervals of approximately 2 m, at a height of 6-9 m, and width of 2 m (see Fig. 1A for example). They are presumed to act as linear features that will guide echolocating bats across roads, above traffic height. At each gantry or underpass, we compared the number of bats using the structure with those crossing unsafely over the road. Where possible, we also compared crossing activity at the gantries and underpasses to that at adjacent or nearby severed but unmitigated commuting routes (as detailed below). The only sites we were unable to compare to nearby commuting routes were underpass B on the A590 and the A69 gantry, as explained below. Photographs are provided in Appendix S1.

The A590 High and Low Newton Bypass (Fig. 1B), opened in April 2008, is a 3.8 km dual carriageway (divided highway) in the Lake District National Park. Two underpasses, a bat gantry (Fig. 1A) and two severed but unmitigated pre-construction commuting routes (Appendix S1) were studied. Underpass A (30 m length ×6 m width ×3 m height) carries a bridleway (a wide footpath designed for horses, pedestrians and cyclists) beneath the road. It is located near to (but not on) a known commuting route, but trees and shrubs were planted along 200 m of the road in an attempt to divert bats from the unmitigated commuting route (on a severed tree-line to the north) that we also surveyed for comparison. Underpass B (30 m length ×6 m width ×5 m height, 500 m south of A) was built to carry a hedgerow-lined minor road, a known commuting route. There were no other previous commuting routes in close proximity for comparison to this underpass. The bat gantry, approximately 1000 m further south, crosses the road at a known commuting route along a severed hedgerow. We also surveyed a second unmitigated commuting route on a severed hedgerow 400 m north of the gantry, where the road now lies in a cutting (roadcut) up to 20 m deep.

The A595 Lillyhall to Parton Improvement, opened in December 2008, is a 5.1 km dual carriageway. A bat gantry was installed where the bypass bisected woodland (Appendix S1). We surveyed this gantry and an unmitigated commuting route 90 m to the north where a hedgerow was severed by the road.

The A66 Stainburn and Great Clifton Bypass, opened in December 2002, is a 4.2 km three-lane carriageway, bisecting a 30 m wide strip of mature woodland. The gantry is located at the western edge of the wood 15 m from the pre-construction commuting route, a bridleway within the wood. We surveyed both the gantry and the pre-construction commuting route. An underpass (15 m length ×5 m width ×2.5 m height) was built at the eastern edge of the wood to carry a small stream, and its use by crossing bats was also investigated.

The A69 Haydon Bridge Bypass, opened in March 2009, is a 2.9 km two-lane carriageway. A bat gantry was constructed at the site of a bat flight line where the road severed a hedgerow (Appendix S1). In pre-construction surveys, minimal bat activity was recorded on all potential commuting routes within 1 km, with the exception of Gee's Wood 800 m east, where the new road bridged a 100 m wide, 10–20 m deep wooded valley with a stream. We therefore conducted surveys at the gantry only.

Survey Methods

Surveys were conducted in June and July, on the A590 in 2010, and all other sites in 2011. Ten 90 min surveys were completed at each crossing point, five commencing at sunset and five starting 90 min before sunrise. Only the five dusk surveys were conducted on the A69 due to low activity levels at dawn. Surveys were conducted on warm, still, dry nights to avoid weather dependent variation in bat activity. At each crossing point an observer was positioned on the verge (grassed bank) either side of the road, equipped with a Pettersson D240× broadband bat detector (http://www.batsound.com, Uppsala, Sweden) and a solid state recorder (Edirol R-09HR, http://www.roland.co.uk, Swansea, UK) set up to automatically detect and record bat echolocation calls (see [1] for details). Since all events were "time-stamped", observations of crossing bats were later matched to echolocation call recordings for species identification. A Pettersson D500× (http://www.batsound.com) bat detector (suitable for automated logging) was also placed in the central reservation (median) when one was present to increase the chances of detection and aid species identification. Both detectors provide recordings that preserve all essential frequency and amplitude information of the echolocation calls, making them the most appropriate choice for species identification. Bats recorded but not observed were excluded, although this was rare. Two observers were used to maximise observations and ensure crossing bats were not missed. All equipment was time synchronised and observers conferred via two way radios. Flight height, direction, distance from the gantry and time of crossing were recorded for each bat. Records were later combined and duplicates removed. Measured points of reference were used to estimate heights and distances to the nearest metre. The bat gantry, fencing and road signs were used



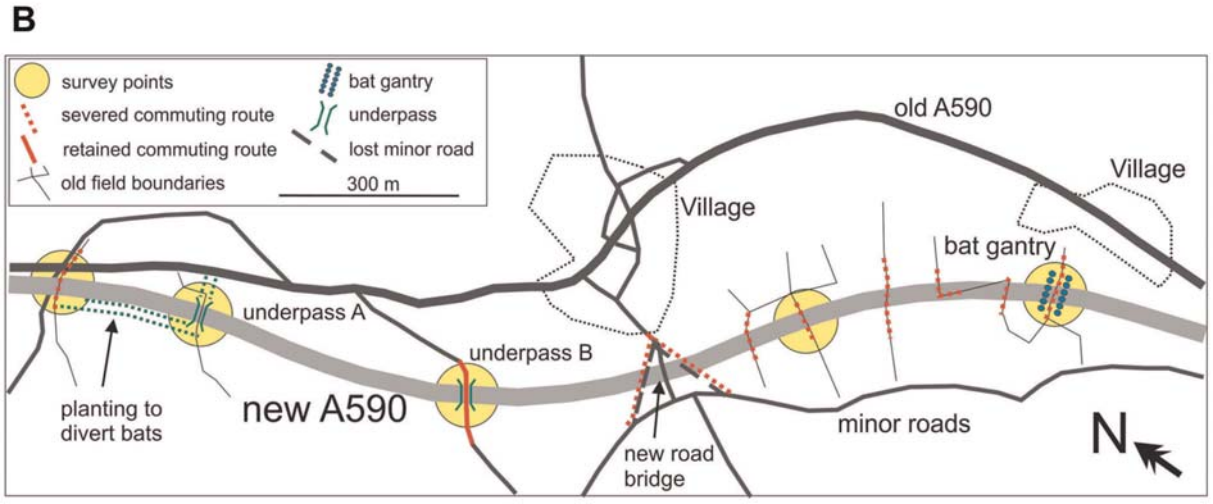


Figure 1A. Photograph of a typical bat gantry (A590). Figure 1B. Map of the study sites on the A590.
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for vertical references, and road markings and crash barrier posts provided horizontal reference points. A clear point of reference was always in view and estimations were made without difficulty. Flight height was recorded over the road, with the majority of bats (87%) crossing at constant heights. For those bats which altered their flight height during crossing (8% decreased height and 4% increased) the lowest flight height over the road was recorded. To corroborate observations night vision digital video cameras (Sony

Nightshot DCR-SR75E and DCR-SR35, http://www.sony.co.uk, Basingstoke, UK) were set up on each verge facing the gantry or commuting route over the road, alongside heterodyne bat detectors used to indicate presence on the recordings (Batbox III, www.batbox.com, Steyning, UK) and infrared lights. However, these were found to be unnecessary with visual observations providing sufficient information.

At the underpasses, these methods were repeated on the road above and an additional observer with the same equipment was positioned at one end of the underpass below. Infrared lights were used to illuminate the underpass, and a night scope (Dedal generation 2, www.nightvision.ru) was used to aid observations. A Pettersson D500× bat detector (http://www.batsound.com) was placed in the centre of the underpass to aid in species identification.

Species Identification

Batsound Pro software (http://www.batsound.com) was used to identify species from sonograms of their calls [32]. In most cases, *Myotis* and *Nyctalus* were identified only to genus because of similarity in call structure [32]. *Myotis nattereri*, *M. mystacinus*, *M. daubentonii* and *M. brandtii* are widespread in the area [33]. *Nyctalus noctula* is widespread, and *N. leisleri* is rare. However, *Nyctalus* data were not analysed as bats flew at heights greater than 15 m over the road and commuting activity was low at most sites. A small proportion (<5%) of *Pipistrellus* calls was classified only to genus level, because of the overlap of call parameters of *P. pipistrellus* and *P. pygmaeus*. *Plecotus auritus* was also present, but will have been under-recorded because of its low intensity echolocation call [32]. Species identification was not reliable for 30% of crossing bats due to noise or low intensity recordings. These records were therefore omitted for species specific analyses. All records (excluding *Nyctalus*) were used in all other analyses.

Definitions

'Safe' and 'unsafe' crossing heights were defined as being greater and less than 5 m above the road surface respectively. The maximum height for heavy goods vehicles in the UK is 4.9 m [31]. Bats crossing the road below 5 m are therefore at risk of collision.

Two estimates of 'use' of the gantry were defined: bats crossing the road within 2 m or 5 m of the gantry at a safe height. These definitions are based on observations from the literature: *Myotis mystacinus* commuting at dusk from a roost to a foraging area flew 0.3–1.7 m from a hedgerow, with the greatest distances recorded only at irregularities in the hedge structure [34]. Commuting *M. daubentonii* flew 3.2–5.8 m from a forest edge and 2.1–4.5 m from a wall [35].

Data Analysis

We have provided statistical analyses, but in some cases, whether or not a particular result was statistically significant contributed little towards assessing the effectiveness of the crossing structures. For example, even if significantly more bats cross a road safely than unsafely, the impact on population trends ultimately depends on the proportion of the population that is killed in collisions. Statistical analyses were carried out using R [36]. Wilcoxon signed ranks tests used the function *Wilcoxsign_test* from the package *coin* [37] to compare activity per survey (n = 5 for A69, n = 10 for all other sites) between underpasses and the road above and between bats crossing at gantries and at unsafe heights below. Each survey was treated as independent. Although activity was generally lower at dusk, there was no observable variation in the behaviour of crossing bats between dusk and dawn and so the data were combined. Some individuals may have been recorded several times during surveys but this was unavoidable, and each crossing event was considered to be important regardless of this. The relationship between flight height and verge height was investigated using Spearman's rank correlation (*cor.test*) and comparisons were made between species using Kruskal–Wallis tests (*kruskal.test*) and pairwise Wilcoxon rank sum W tests (*wilcox_test* function, package *coin*) with Bonferroni corrections. On the A66, observations were made across the entire 30 m section of severed woodland that included both the gantry and the severed commuting route. The heights and positions of all crossing bats were used to generate a kernel estimate of crossing intensity, using the *density* function in the *spatstat* package [38].

Results

Underpasses

At underpass A on the A590, activity was low (Table 1, Fig. 2A), but 69% bats preferred to fly over the road rather than use the underpass ($Z = -2.39$, $P = 0.03$). Of bats crossing the road, 88% did so at unsafe heights. *Pipistrellus pipistrellus* and *P. pygmaeus* were detected in the underpass. *P. pipistrellus*, *Myotis* and *Plecotus auritus* were detected flying over the road. Over the same period, more bats crossed the road at the nearby unmitigated, severed commuting route (Table 1, Fig. 2A), 58% at unsafe heights. Most were *P. pipistrellus* and *P. pygmaeus*, approximately half crossing below 5 m. Four of the five *Myotis* detected crossed below 5 m. No bats were observed flying along the planted diversion to the underpass, but observers were only able to monitor this where it left the original commuting route.

Activity levels were higher at underpass B than at A (Table 1, Fig. 2B) and many more bats (96%) used the underpass than flew over the road above at unsafe heights ($Z = 2.80$, $P = 0.002$). *P. pipistrellus*, *P. pygmaeus* and *Myotis* were all detected both in the underpass and over the road.

Activity in the underpass below the A66 was low (Table 1, Fig. 2C), with only 4% of bats crossing through it, in comparison to 60% crossing at unsafe heights over the road above ($Z = 2.80$, $P = 0.002$). *P. pipistrellus*, *P. pygmaeus* and *Myotis* were detected over the road and in the underpass.

Bat Gantries

At all sites, few bats crossed using the gantry (Fig. 3). At the A590 gantry, four times as many (41%) crossed the road at unsafe heights as crossed within 2 m of the gantry (11%; $Z = 2.61$, $P = 0.008$), and 1.4 times as many as crossed within 5 m of the gantry (30%; $Z = 1.49$, $P = 0.15$). At the A595 gantry (Fig. 3B), far more bats (84%) crossed the road at unsafe heights than flew within 2 m (<1%; $Z = 2.81$, $P = 0.002$) or 5 m (6%; $Z = 2.81$, $P = 0.002$) of the gantry. At the A69 gantry (Fig. 3C), more bats crossed the road at unsafe heights (17%) than flew within 2 m of the gantry (8%; $Z = 1.17$, $P = 0.31$), but fewer bats crossed at unsafe heights compared to those flying within 5 m of the gantry (42%; $Z = -2.14$, $P = 0.06$).

At the A66 survey area (including both the gantry and the pre-construction commuting route, Fig. 3D), far more bats (70%) crossed at unsafe heights, than flew within 2 m (2%; $Z = 2.81$, $P = 0.002$) or 5 m (9%; $Z = 2.81$, $P = 0.002$) of the gantry. The kernel density estimation for the A66 (Fig. 4) shows a high concentration of bats crossing at unsafe heights centred at the unmitigated pre-construction commuting route, and low activity around the gantry.

On the A595 the number of bats crossing at the nearby unmitigated, severed commuting route (Fig. 3) was comparable

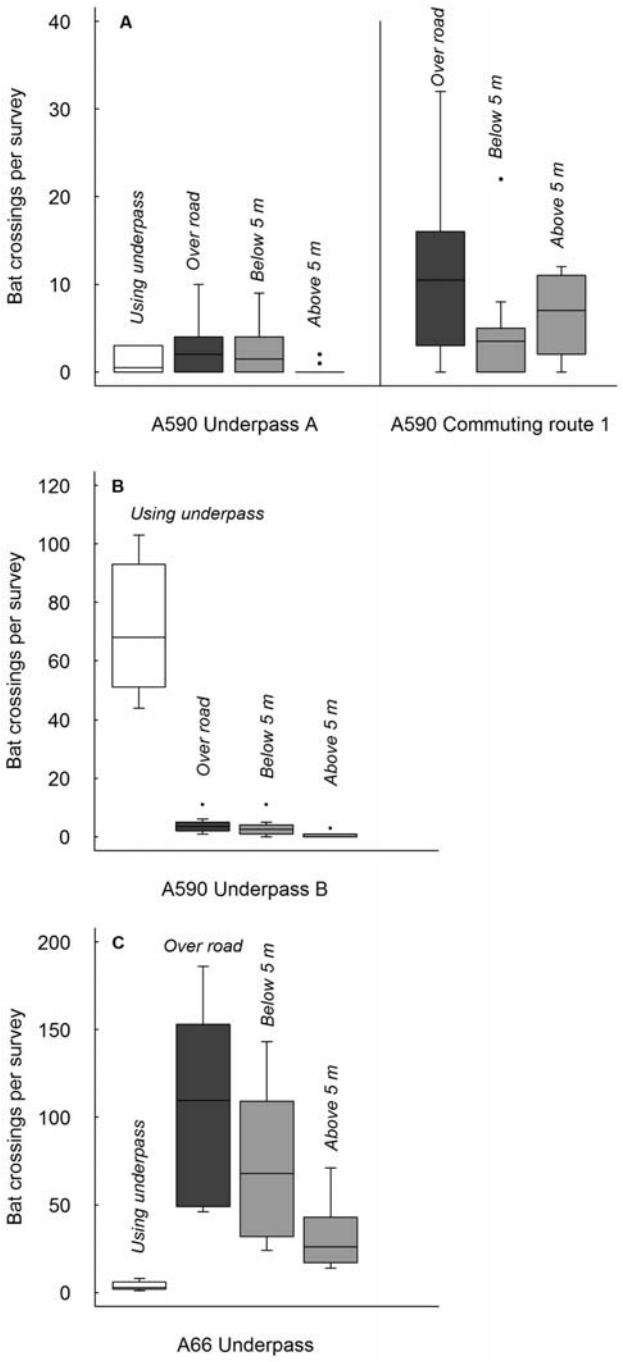


Figure 2. Boxplots of the number of bats crossing per survey at each underpass. Boxplots (median with upper and lower quartiles) for the number of bats crossing per survey (n = 10) at each underpass (numbers crossing using underpass, over the road above and at safe and unsafe heights over the road), and at the unmitigated commuting route on the A590 which was diverted to underpass A (numbers crossing over the road and at safe and unsafe heights).
doi:10.1371/journal.pone.0038775.g002

with that crossing in the vicinity of the gantry, and 94% of bats crossed at unsafe heights.

At the unmitigated, severed commuting route near the gantry on the A590 foraging activity of *P. pipistrellus* was observed during all surveys on the western side of the road, but only 19 bats crossed

Table 1. The crossing behaviour of all bats from all surveys for each study site (NB the number of bats crossing at safe heights at the gantries includes those 'using' the gantry).

Road	Site	Total crossing	'Using' gantry (within 2 m)	'Using' gantry (within 5 m)	Using underpass	Unsafe height over road	Safe height over road
A590	Underpass A	36	-	-	11	22	3
	Commuting route 1	113	-	-	-	65	48
	Underpass B	904	-	-	864	32	8
	Bat gantry	104	11	31	-	43	61
	Commuting route 2	19	-	-	-	0	19
A595	Bat gantry	96	1	6	-	81	15
	Commuting route	77	-	-	-	72	5
A66	Combined survey area	1117	24	100	39	751	327
	(Bat gantry, Commuting route & underpass)						
A69	Bat gantry	65	5	27	-	10	55

doi:10.1371/journal.pone.0038775.t001

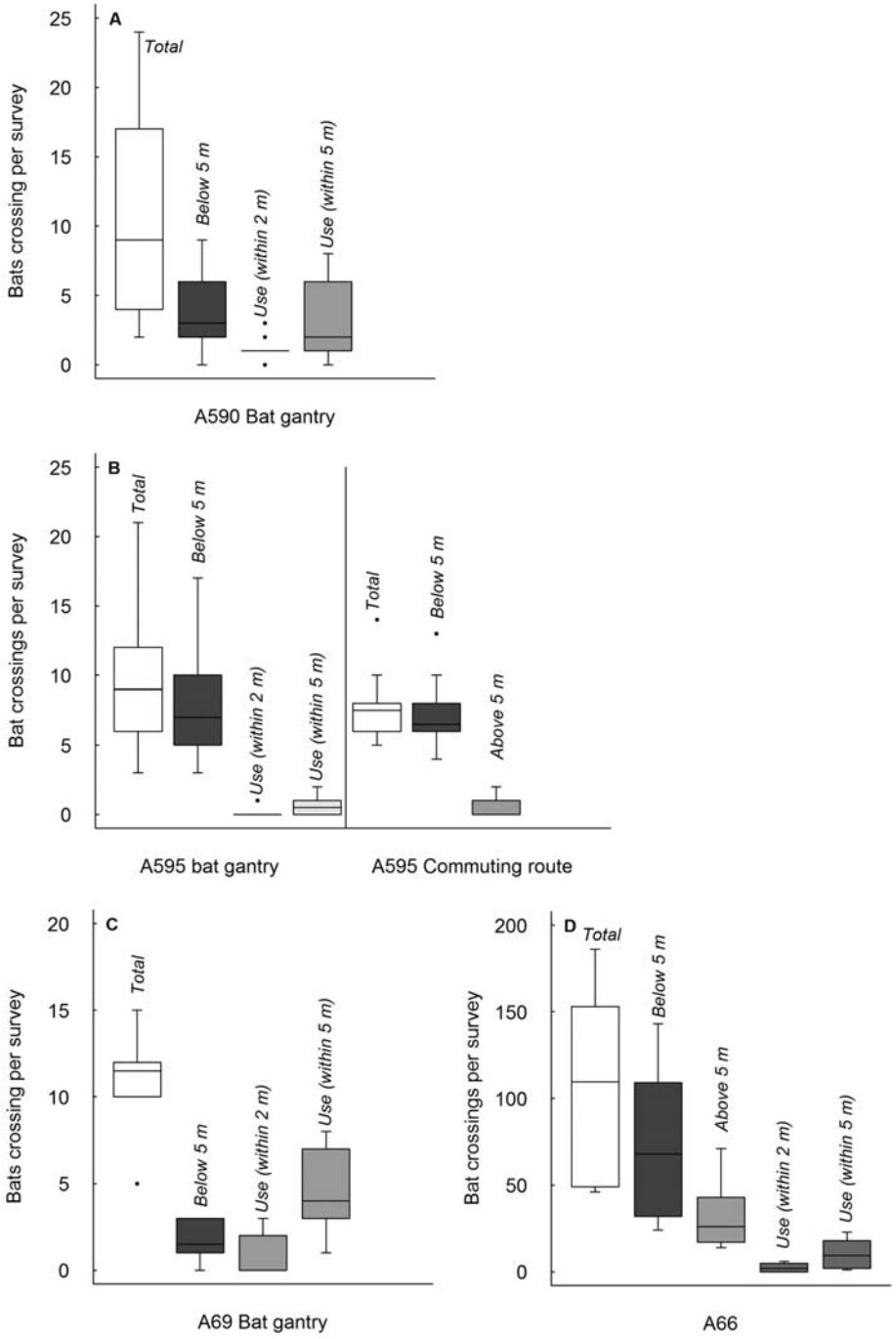


Figure 3. Boxplots of the number of bats crossing per survey at each bat gantry. Boxplots (median with upper and lower quartiles) of the number of bats crossing per survey (n = 10) at the four bat gantries, together with data on total number crossing, the numbers crossing at safe and unsafe heights, numbers 'using' the gantry according to both estimates (within 2 and 5 m), and the numbers crossing at nearby, unmitigated, severed commuting route nearby.
doi:10.1371/journal.pone.0038775.g003

over 10 surveys, all at safe heights. Although other severed commuting routes (shown in Fig. 1) were not surveyed, significant crossing activity was not observed during reconnaissance.

The Influence of Verge Height

The mean crossing height of all bats across all sites (excluding underpasses) was positively correlated with verge height (estimated to nearest 0.5 m) at the point of crossing (Spearman's rank; $r = 0.34$, $n = 1552$, $P<0.0001$). This correlation was significant at the species/genus level, with *Myotis* showing the strongest relationship (*Myotis*: $r = 0.46$, $n = 55$, $P<0.001$; *P. pipistrellus*: $r = 0.40$, $n = 284$, $P<0.0001$; *P. pygmaeus*: $r = 0.34$, $n = 343$, $P<0.0001$). Crossing height above the height of the verge was found to vary between genera (Fig. 5). No difference was found between the two

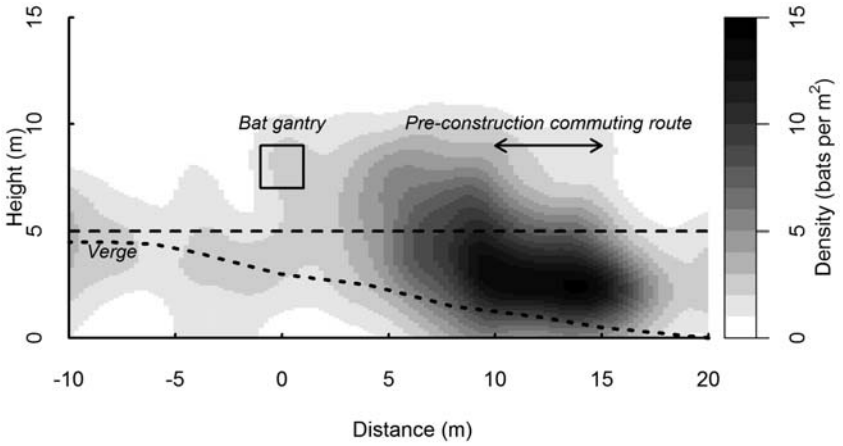


Figure 4. Kernel intensity estimation of the density of crossing bats across the A66 site. Gaussian kernel and bandwidth of 1 m used (n = 1078). The section of severed woodland at the A66 site is shown. The gantry is located at distance 0 m (height marked by square), and the pre-construction commuting route at 10–15 m. 'Unsafe' crossing heights are located below the dashed line. The dotted line marked verge shows the decrease in verge height above the road from left to right.
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Pipistrellus species (Wilcoxon rank sum; W = 47193.5, $P>0.05$ after correction), but *Myotis* flew significantly lower than both *P. pipstrellus* (Wilcoxon rank sum; W = 5306.5, $P<0.0005$ after correction) and *P. pygmaeus* (Wilcoxon rank sum; W = 5935, $P<0.0001$ after correction). Only three *P. auritus* were detected, and all crossed below the height of the verge at <3 m over the road.

Discussion

This is the first study to assess the effectiveness of road crossing structures for bats, by measuring the proportion of individuals that used these structures to cross safely. Although a limited study of such diverse structures cannot be definitive, we believe it demonstrates that some current practices are failing. We found no evidence that bats used gantries in preference to nearby, severed but unmitigated commuting routes. At all but one site (A69, where activity was low), the majority of bats crossed at unsafe heights, even in proximity to gantries. Of seven mitigation structures studied, only one underpass was effective in carrying the majority of bats safely across the road.

Underpasses

Underpass A on the A590, and the A66 underpass, are not effective mitigation measures: very few bats flew through them relative to the number crossing at unsafe heights over the road above, and in the case of underpass A, at an original commuting route nearby. Underpass B on the A590 showed high levels of use by commuting bats, with just 4% crossing at risk of collision mortality on the road above. This underpass is effective in allowing bats to cross the road safely. However, the lack of robust pre-construction population data makes it difficult to assess how effectively this underpass can protect bat populations. Even though a high proportion of bats use the underpass, if bat populations have declined since construction and the road acts as a barrier, then the underpass becomes ineffective. Nevertheless, underpass B preserved a pre-construction commuting route, with no necessity for commuting bats to alter their flight course or height. Although replication is needed, this shows that underpasses can be effective when built over existing commuting routes. This makes sense in the context of the high fidelity that bats show to their commuting routes [39,40,41]. Underpass A and the A66 underpass were unsuccessful probably because they require commuting bats to alter their course and flight height. Both underpasses are also lower than underpass B, but several studies report bats flying through even smaller structures e.g. [24].

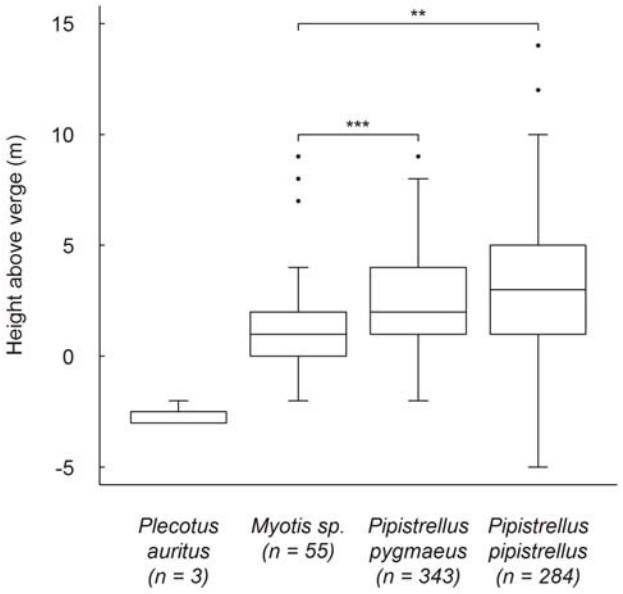


Figure 5. Boxplot of flight height above verge height of identified crossing bats at all sites. Median with upper and lower quartiles. Significant differences shown for *Myotis* and *Pipistrellus* species ** $P<0.0005$, *** $P<0.0001$. Excludes underpass sites. Verges are elevated on either side of the road and are above road height, therefore negative values indicate bats flying across the road below the height of the verge.
doi:10.1371/journal.pone.0038775.g005

Bat Gantries

Bats did not cross at gantries more than at unmitigated road crossings, and gantries did not effectively increase the height at

which bats flew above the road. There was no evidence that bats were 'using' gantries by flying in close proximity to them, as they do along hedges [34,35].

These bat gantries are failing to perform the function for which they were built, even at well-established sites such as the A66, where the gantry has been in place for nine years and is only 10–15 m from the original commuting route. Although road kill counts were not performed, it is well documented that bats are killed on roads in high numbers [6,7,8,9] and mortality may be high enough to be unsustainable [23].

Verge Height

The strong correlation between verge height and the average crossing height of bats suggests that increased verge height may have some potential in raising flight height above traffic. This effect was found to vary between species: *Myotis* species were most sensitive to changes in verge height and flew closer to the verge than *Pipistrellus* species, as did the few *P. auritus* observed. However, increased verge height generally widens the open terrain that must be crossed (since they are inclined away from the road, higher verges are usually wider, see photographs in Appendix S1), which could deter some species from crossing, increasing the barrier effect. Very few bats crossed the road at the second unmitigated commuting route on the A590, where verges are 20 m high, and the width of the open space is 80 m. Similarly, in Germany, *M. bechsteinii* were observed to frequently fly over a two lane road with a connecting tree canopy, but not over a four lane motorway with a gap in the forest [2].

Habitat Continuity

It has been suggested that crossing structures will be more effective if continuous with the vegetation on either side of the road [8]. However, even though the A66 gantry is connected to mature woodland on either side and is only 10–15 m from the commuting route, it is still ineffective. Commuting bats use linear habitat elements not just for navigation, but also to obtain protection from predation and wind and as foraging microhabitats [29,42]. More substantial structures that provide shelter and perhaps bear a closer resemblance to natural features are likely to be more successful, for example a planted green bridge that provides a continuation of hedgerow, or tree lines over the road. Green bridges, although built for other wildlife, are only just being considered as mitigation measures for bats, and evidence is still needed for their effectiveness. Ten species of bat were found to use green bridges in Germany, with higher use than conventional road bridges, but results focussed on bats using the structures and did not look at those crossing the road below [25]. A simpler (but as yet untested) alternative that may be practical and effective on narrower roads is the 'hop-over': mature trees that overhang the road so that their crowns bridge the gap above the road [43].

Species – Specific Effects

Nyctalus species do not appear to be adversely affected by roads. High foraging activity was observed over traffic at one site (A69), and small numbers of commuting *Nyctalus* were observed over the A590 at heights of over 15 m above the road. In other studies *Nyctalus* species have been observed flying high over roads with no recordings in underpasses [1,2,24], and low incidences of collision mortality [6,7]. Roads have also been found to have less of an impact on habitat use by species that also forage in more open habitat, such as *Barbastella barbastellus* [2].

All other species detected in this study crossed at unsafe heights over the road. Differences in crossing heights were found between species, with *Myotis* species (and the three detected *P. auritus*) flying lower over the road than *Pipistrellus* species, increasing their vulnerability to collision mortality.

The Effectiveness of the Survey and Monitoring Process

The bat gantries and one of the underpasses were installed because they were believed to be on significant commuting routes. However, we found activity was low at all gantry sites with the exception of the A66. Either activity has greatly diminished post-construction, adding to the conclusion of mitigation failure, or the assessment of these sites as major commuting routes was perhaps flawed. The absence of robust pre-construction monitoring means that we cannot say which explanation is correct. This raises serious questions about the effectiveness of the survey, assessment, mitigation and monitoring process. Several of the structures we have shown to be ineffective were said to be working in the commissioned reports [22], using the criterion that bats were seen to use them. Are other aspects of the reports equally flawed? Limited resources are available for conservation and it is vital that they are used effectively. Failure to do so makes no contribution to conservation and alienates further those outside conservation who question the use of public funds on conservation, e.g. the recent spending of £0.5 M on bat gantries [44,45].

Conclusions and recommendations

We assessed only a small number of mitigation structures, but the results are sufficiently striking that wider appraisal is essential if mitigation against road construction is to be effective. Wire bat gantries, of the type studied, should not be used, and attempts to divert original commuting routes should, if possible, be avoided. Underpasses built on existing commuting routes can be effective crossing structures, if commuting bats can maintain their original course and flight height. Further investigation into more substantial, natural crossing structures over roads, such as green bridges, and simpler options such as tree 'hop-overs', is needed. Unique aspects of individual sites, such as tree cover, hedges and topography must be exploited to make mitigation solutions as natural as possible and appropriate to the bat species present. Robust and comparable pre- and post-construction monitoring must be carried out that assesses more objectively the need for mitigation and its effectiveness.

Supporting Information

Appendix S1 Photographs of study sites.
(ZIP)

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Author Contributions

Conceived and designed the experiments: AB JA. Performed the experiments: AB. Analyzed the data: AB. Wrote the paper: AB JA.

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WC1060 DEVELOPMENT OF A COST-EFFECTIVE METHOD FOR MONITORING THE EFFECTIVENESS OF MITIGATION FOR BATS CROSSING LINEAR TRANSPORT INFRASTRUCTURE

Final report 2015

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Executive Summary

Background

Recent research has shown that roads can have negative impacts on bats with landscape scale reductions in bat activity and diversity (Berthinussen & Altringham 2012b), reduced reproductive success (Kerth & Melber 2009), and mortality through collisions with traffic (Lesinski 2007, Lesinski et al. 2010, Russell et al. 2009). As all UK bat species are protected by European law via the Habitats Directive, they must be considered in the Environmental Impact Assessment (EIA) of developments such as roads, and any unavoidable adverse impacts must be mitigated.

EIA is an important aspect in the planning application process for large scale developments. Developers must prepare and submit Environmental Statements detailing the impact of the proposed development on the environment, including protected species such as bats. Environmental Statements should be prepared in an objective and systematic way and provide sufficient information for decision-making and to inform the mitigation process. They should also provide baseline activity data for the species in question which can be compared against data collected post-construction to draw conclusions about the effect of the development and the effectiveness of mitigation. However, in practice the quality and quantity of data are often insufficient, with poor experimental design and a clear lack of objectives, making it difficult to draw conclusions.

Little evidence exists for the effectiveness of currently used mitigation structures for bats on roads, such as underpasses, bridges and wire gantries. Recent reviews of case studies of bat mitigation in the UK found that most reports were qualitative and inconclusive (Altringham 2008, O'Connor et al. 2011, Berthinussen & Altringham 2012a), and failed to make the important distinction between the use of structures by individuals and the effectiveness of structures in maintaining local population sizes. Without this information it is impossible to know whether mitigation measures are effectively protecting local bat populations from adverse effects, as required by law. Reliable methods that can be used in EIA and proven robust in identifying effective bat mitigation are needed to ensure a more efficient development process and more positive outcomes for bats.

Aims

The aims of the project were to develop standardised cost-effective survey methods to assess the effect of linear transport infrastructure on bats (on a landscape scale) and the effectiveness of mitigation measures such as crossing structures (on a local scale). The focus is on road construction but rail has also been considered as another form of linear infrastructure. The methods were to be quantitative, robust, replicable, appropriate for statistical analysis and the comparison of pre, during and post-construction data, and suitable for providing baseline data for Environmental Statements. Clear protocols were to be created for use by conservation practitioners and the consultancy industry. The effectiveness of some currently used mitigation measures were to be assessed while testing the methods, and protocols were to be produced to inform best practice, management and appropriate mitigation solutions for future road (and rail) developments.

Survey methods

We developed protocols that we believe provide an acceptable compromise between providing robust data from which to draw conclusions and cost-effectiveness. We determined the minimum number of surveys that can be conducted per site to provide sufficient data for analysis and we tested the effect on statistical power of reducing survey duration for cost-effectiveness.

Two survey protocols were developed to address the two different aspects specified in the aims:

1. A landscape scale survey method using transects to assess the effect of linear infrastructure on bats at a population level

2. A local scale observational survey method to assess the effectiveness of mitigation measures such as crossing structures over or under linear infrastructure

We stress that these methods should be used in conjunction with existing practices: desk studies and fieldwork to identify known roosts, foraging areas and commuting routes, particularly of rare and vulnerable species. The identification of points where bats cross the proposed road is vitally important, since it is in these areas that mitigation efforts must be concentrated.

The landscape scale transect protocol measures bat activity and species diversity over a large area surrounding the linear infrastructure, and uses statistical modelling to detect changes in bat activity and diversity related to its presence. A reduction in bat activity and/or diversity in proximity to the infrastructure indicates the strong likelihood of a negative impact on nearby bat populations. Pre-construction surveys provide baseline data, and can be compared with data collected during and post-construction to detect overall changes. The transect protocol will also deliver data suitable for habitat suitability modelling.

The local scale observational protocols are more straightforward, and provide a simple and effective method for objective assessment of mitigation structures. Set definitions and criteria are used, and the effectiveness of crossing structures are assessed by comparing the number of bats using the structure to cross, with those using the original pre-construction commuting route, and those crossing at risk of being killed by traffic.

We tested the landscape scale transect protocol successfully at seven road sites (three established single carriageway A roads, an established dual carriageway A road, an established motorway, a motorway during construction and a motorway immediately post-construction) and two rail sites (both established). We found that motorways (established, pre- and during construction) had a consistent negative effect on bat abundance and diversity. The effects of A roads were inconsistent with effects noted on two of the three single carriageways studied but not on the dual carriageway, and further research is needed to explain these differences. These findings corroborate previous research (Berthinussen & Altringham 2012b), showing that roads can have large scale negative impacts on bat activity, although more research and population modelling is necessary to determine how these effects influence population sizes and viability. One of the two railway sites was found to have a negative effect on bat activity and diversity, suggesting that railways can have similar impacts on bats as roads, but further research is needed.

We also assessed the effectiveness of nine existing mitigation features for bats on roads (three underpasses, three bat wire bridges, an overpass, an environmental bridge and a green bridge). We found one of the underpasses and the green bridge to be effective in guiding a large majority of bats under or over the roads. We concluded that underpasses were more likely to be used successfully by commuting bats than overpasses and bat gantries/wire bridges, both of which were consistently ineffective. It is worth noting that a bat gantry erected close to a known commuting route was not being used by bats nine years after construction (Berthinussen & Altringham 2012b). Green bridges appear to have considerable potential as mitigation structures, but more research is needed. Although the sample sizes are small, and further research is needed to draw firmer conclusions, the results suggest that the effectiveness of crossing structures increases with their size, connectivity and similarity to natural linear features.

In addition, a radar feasibility study was also conducted by the Animal and Plant Health Agency (APHA) to test, for the first time, the use of horizontally scanning X-band radar to track bat movements across large areas of the landscape, with the aim of studying their responses to road, rail and other linear infrastructure. Initial results are promising, suggesting that the technique can track small bats close to linear infrastructure, but that use as an effective survey tool will require further development and it will not be suitable in all situations.

Reporting

We have provided and discussed the results from all sites at which we tested our survey methods. We have provided full survey protocols for both methods with a rationale, technical advice on data format, organisation and storage, and full instructions for data analysis. Example datasheets and spreadsheets are provided as well as step-by-step instructions for use with the suggested software. We also provide and explain criteria for drawing conclusions about the effects of linear infrastructure and the effectiveness of mitigation measures. Based on our findings and those of previous studies we provide advice for improving on current mitigation techniques.

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1. Background and rationale

Habitat fragmentation is widely accepted to be one of the most important factors in the loss of biodiversity. Roads and railways, by their very nature as extensive linear networks, have the potential to fragment the landscape and prevent the free movement of species. The detrimental effects of roads on biodiversity are becoming increasingly well-recognised (e.g. Forman *et al.* 2003) and it has been known for some time, from work on birds, that flying animals are not immune to these effects (e.g. Reijnen & Foppen 1994; Reijnen *et al.* 1996). It was therefore assumed that bats would also be affected, particularly given the reliance of many species on linear landscape features such as treelines and hedgerows for commuting (Frey-Ehrenbold *et al.* 2013), since these are often severed by road and rail. It has been shown that bats are indeed reluctant to cross roads (Kerth & Melber 2009; Russell *et al.* 2009; Bennett & Zurcher 2013), that the presence of a motorway can reduce reproductive success by reducing habitat quality (Kerth & Melber 2009), that roads kill bats in significant numbers (Lesiński 2007; Russell *et al.* 2009; Lesiński *et al.* 2010, Medinas *et al.* 2013) and that mortality rates are in many cases likely to be unsustainable (Russell *et al.* 2009). Hotspots for mortality are found where flyways intersect with roads (Lesiński 2007; Russell *et al.* 2009). There has been little research into the effects of railways on both bats and other wildlife, and given the extensive network of rail systems in many countries it is important that rail effects are also assessed objectively. A literature review on "Bats, roads and railways" can be found in Appendix A.

All bats are strictly protected across the European Union (EU), including the United Kingdom, via the Habitats Directive. The Conservation of Habitats and Species Regulations 2010 (known as the Habitats Regulations) transpose the EU Habitats Directive into domestic law. If a development affects bats, it may be possible to avoid harm or minimise impacts via agreed method statements or by imposing restrictions on how, when or where the works are carried out. In such circumstances the specified works should be controlled by the local planning authority through the imposition of appropriate planning conditions. In most cases, a mitigation and compensation strategy would need to be put in place to ensure that favourable conservation status is not adversely affected by the development (including for example, creation of new habitats to offset any damage). A licence may be required if the works would otherwise result in offences under the Regulations (usually for damage or destruction of roost sites). Licences will only be issued if the licensing authority is satisfied that the proposed development meets the three tests set out in the Habitats Directive which are: that it fulfils one of the purposes set out in the Directive; that there is no satisfactory alternative; and that it will not be detrimental to the maintenance of the population of the species concerned at a favourable conservation status in their natural range.

A number of mitigation strategies have been developed to reduce the influence of roads on bats. These are designed to make roads easier and safer for bats to cross by maintaining commuting routes severed by the roads. Continuity of these flightlines is maintained by retaining hedges and trees as close to the road as possible, and by building underpasses, overpasses or wire bridges (or gantries). Road lighting schemes are also designed to keep commuting routes as dark as possible, since it is known that many species avoid light (Stone *et al.* 2009; Stone *et al.* 2012). Light may also be used to try to deter bats from crossing at particularly dangerous locations. More general mitigation and compensation may be carried out, through tree planting and pond digging, to replace habitat lost to development. However, little evidence exists for the effectiveness of these mitigation strategies in protecting bat populations.

Several studies have shown that a wide range of bat species use underpasses to cross beneath roads (e.g. Bach *et al.* 2004; Kerth & Melber 2009; Boonman 2011; Abbott *et al.* 2012a; Berthinussen & Altringham 2012a). However, occasional use by an unknown proportion of bats does not guarantee either habitat accessibility or safe crossing routes for the bat population as a whole. For example, despite the existence of three underpasses along 5 km of motorway that bisected a forest, Bechstein's bats, *Myotis bechsteinii,* rarely used them and lost access to important roosting and feeding habitat (Kerth & Melber 2009). Abbott et al. (2012) found that almost a third of road-crossings by lesser horseshoe bats, *Rhinolophus hipposideros,* were directly over a motorway, despite their regular use of three underpasses along a 1 km stretch. It cannot be

assumed that the reluctance of forest-adapted bats to fly in the open means that they will take detours to use underpasses: many bat species appear reluctant to deviate from their original flight paths after road severance (Kerth & Melber 2009; Abbott 2012; Berthinussen & Altringham 2012a). Conventional overpasses carrying minor roads or footpaths do not appear to be effective crossing points for bats, relative to routes under roads (Bach *et al.* 2004; Abbott *et al.* 2012a) and wire gantries or bridges tested to date (up to nine years after construction) are ineffective (Berthinussen & Altringham 2012a). Land or green bridges (wide, vegetation-covered bridges) are likely to be more effective given their similarity to natural bat commuting features, but are potentially more expensive than other structures and have yet to be tested.

The success and cost-effectiveness of such mitigation is the subject of media interest and recent parliamentary questions have queried the cost-effectiveness of structures designed to help bats cross roads. In principle, the success and cost-effectiveness of mitigation are assessed by monitoring against clearly formulated goals. In practice however, data quality and quantity have rarely been sufficient to make an objective assessment of either, in part due to poor experimental design and a lack of clear objectives. The lack of proven methods can hamper development and can result in additional costs, e.g. when expensive retrofitting of new structures is deemed necessary. Recent work has brought a more rigorous and analytical approach to the subject, demonstrating directly, for the first time, that roads can have a major, negative impact on bat activity and diversity (Berthinussen & Altringham 2012b) and that the most widely adopted mitigation measures are not effective and therefore not cost-effective (Berthinussen & Altringham 2012a). A reliable method that can be used in environmental impact assessment and proven robust in identifying effective bat mitigation would ensure a more efficient development process and more positive outcomes for bats.

The aims of the project were to develop and validate bat survey techniques for pre-, during and post-construction of linear infrastructure, and to set out clear protocols for a standardised survey in sufficient enough detail to allow it to be applied consistently by conservation practitioners and the consultancy industry. The methods were to be quantitative and appropriate for statistical analysis. The project also required the methodology(s) to be applied at a sufficient number of sites to allow a robust assessment of the success and cost effectiveness of different mitigation measures for bats on roads. The project was to focus on road construction, but also to consider the implications of rail.

The project contributes to the fulfillment of three important commitments of government and transport agencies in relation to the natural environment:

- the need for the creation of coherent and resilient ecological networks (identified by the Natural Environment White Paper in England)
- the need to improve evidence-based decision making in conservation and other areas of policy and practice
- the maintenance and restoration of European Protected Species to favourable conservation status

The project was carried out by a team from the University of Leeds, in collaboration with APHA (Animal and Plant Health Agency) for the radar work.

6. Discussion

We have developed methods that are capable of detecting changes in bat activity and diversity in proximity to linear transport infrastructure on a landscape scale, and that allow for the objective assessment of the effectiveness of mitigation structures for crossing bats on a local scale. As specified in the original aims, the methods are quantifiable, robust, replicable, appropriate for statistical analysis, and can be used to compare pre, during and post-construction bat activity as well as provide baseline data. We have tested the landscape scale method at seven road and two rail sites and the local scale method at nine road mitigation sites. The methods have been refined by reducing survey number and duration, to reach an acceptable compromise between gathering robust data from which to draw conclusions and cost-effectiveness. The statistical analysis methods have also been refined and tested to ensure that they are as accurate and robust as possible. Analysis is necessarily quite complex, but we have made it as simple as possible and provided detailed instructions and code for use in R (R Development Core Team, 2006).

The only significant practical issues we encountered were difficulty in obtaining access to some sites (such as road verges along dual carriageways, roads under construction or transect routes through private land) within the timescale of the project. However, these issues should be overcome by planning well in advance, incorporating bat surveys into the scheme from the outset and liaising with landowners, road agencies and construction companies from an early stage.

In addition to our survey methods, a radar feasibility study was also conducted by the Animal and Plant Health Agency (APHA) to test, for the first time, the use of horizontally scanning X-band radar to track bat movements across large areas of the landscape, with the aim of studying their responses to road, rail and other linear infrastructure. Initial results are promising, suggesting that the technique can track small bats close to linear infrastructure, but that use as an effective survey tool will require further development and it will not be suitable in all situations. See Appendix J for a full discussion of the results.

6.1 Interpreting the results

It is important not only to have robust and standardised survey protocols, but also for the interpretation of results to be objective. We have suggested standardised criteria from which to draw conclusions from the results for each survey method in our protocols (see Appendices D and F). Although statistical significance is useful for identifying and confirming the reliability of differences or relationships, it is not always of critical importance. For example, even if significantly more bats cross a road safely than unsafely, the impact on population trends ultimately depends on the proportion of the population that is killed in collisions. At present, the demographic information is not available to calculate the magnitude at which a negative effect becomes detrimental to a bat population, or what mortality rate is sustainable. Further studies, with appropriate modelling could provide some answers. In the absence of this information, we have based our precautionary advice on the fact that bats are long lived, have low reproductive rates and high infant mortality, and are therefore slow to recover from population crashes. Even small increases in annual mortality or a reduction in reproductive output will have profound effects on local colony size and overall population size (Sendor & Simon 2003; Papadatou *et al.* 2011).

For the assessment of mitigation structures, we suggest that up to 10% of bats may cross the road at risk of collision with traffic, but once this is exceeded, mortality rates are considered to be potentially detrimental to local bat populations. As explained above this is a precautionary figure. Although not all bats crossing at risk of collision will be killed, there is the potential for mortality rates to increase by 10% within a population. Schorcht et al. (2008) modelled the effect of a highway in Germany on a *R. hipposideros* nursery colony and results indicated that an additional mortality of 5 - 7% of this female population caused by the road would lead to colony extinction. We also suggest that for a crossing structure to be effective, at least 90% of crossing bats must use it to fly safely over or under the road. We suggest using the more conservative definition of 'use' (bats flying at a safe height within 5 m of the crossing structure rather than 2 m) which

encompasses bats flying over a relatively wide area. If bats *are* using the structure to navigate across the road then 90% should be an achievable target.

If a majority of bats are 'using' the structure but the suggested threshold is not met, then this suggests that the structure *could* be effective with some improvements. A concerted effort to improve structures should be made in these situations and the effects carefully documented to inform future mitigation. However, until the structure has been improved and the thresholds met, the structure cannot be considered to be effective and may lead to local population decline.

For landscape scale effects, we suggest that if the decline in activity in proximity to the road/railway becomes statistically significant or greater than 20%, then detrimental effects are assumed. As above, this is based on a precautionary principle and related to the ability to detect change in intrinsically highly variable foraging activity patterns in bats. If populations are to be maintained, any statistically significant decline related to the presence of a road has the potential to affect populations. Because of the highly variable nature of bat activity (due, for example, to behavioural changes), even large changes may be hard to detect statistically. We therefore suggest, as a precautionary measure, that a decline of 20%, irrespective of statistical significance, is a cause for concern. Landscape scale changes in activity do not translate directly into increased mortality, but will reflect more complex indirect effects on local bat colonies, such as reduced reproductive success, increased competition following displacement or moves to sub-optimal roosts and foraging areas.

NB. Although we give guideline figures every case should be considered carefully in relation to the local and national scarcity of the species at risk, proximity of the site to major roosts, the proportion of bats using particular flight lines and other considerations.

6.2 Landscape scale effects

Road effects

We detected changes in total bat activity that were correlated with proximity to roads at five of our seven road sites. A negative effect on bat activity was found along single carriageway roads (A487 and A134), an established motorway (M5) and motorways both immediately after construction (A1 Dishforth to Leeming) and during construction (A1 Leeming to Barton). All five of these sites showed substantial changes in bat activity and diversity with distance from the road (with statistical significance at all but one site, the A134). This corroborates our published work on the M6 (Berthinussen & Altringham 2012b), showing that roads can have large scale negative impacts on bat activity that are likely to reflect change in population size. An effect of this scale is likely to be due to a combination of a barrier effect and increased mortality due to roadkill causing a decline in local bat populations (as discussed in Berthinussen & Altringham 2012b). With an effect found on single carriageway A roads, it is possible that minor roads (such as B roads) may also have a negative impact on bats. However, at present there is no evidence for this and further research is needed.

Our results highlight the importance of carrying out species-specific analyses. Road effects on more abundant species, such as *P. pipistrellus,* may mask the effects on less abundant species when total bat activity is modelled, and species with opposing effects, such as *Nyctalus*/*Eptesicus* species may confuse interpretation of the model even further.

We advise that species-specific models are created for all species (with sufficient abundance) and *Nyctalus*/*Eptesicus* species should be removed from the data for the analysis of total bat activity. Our results and previous research show that these species are unlikely to be negatively affected by linear transport infrastructure such as roads, and may even be positively affected in some cases. *Nyctalus* species are known to fly high and forage in open spaces (e.g. Kronwitter 1988). We frequently observed them foraging high over the road surface suggesting they may be attracted to the open spaces above roads to feed. Removing these species from our data had dramatic effects on the road effects detected at two of our sites. On the A40, although model estimates remained non-significant, the road effect changed from positive to negative, and on the A134 the negative road effect was predicted to increase almost five-fold with the exclusion of

these species. At all other sites, the abundance of *Nyctalus*/*Eptesicus* species was low and removing them from the data had no impact on the road effects found. However, it is still important to carry out species-specific analyses for *Nyctalus/Eptesicus* as road effects may not always be neutral or positive. Unexpectedly high numbers of *Nyctalus noctula* were found in counts of road kill along a forested road in Poland (Lesiński *et al.* 2010).

When activity was high enough for individual analyses, we found that *P. pipistrellus*, *P.pygmaeus* and *Myotis* species all had reduced activity in proximity to the road at those sites where total bat activity was affected (with the exception of the A40). Although the magnitude of the road effect varied between species, we cannot make direct comparisons between them since model estimates and predictions may not be exact for those with a low occurrence. However, it is likely that gleaning woodland specialist species such as *Myotis* and *Plecotus* will be more affected than generalist species such as *P. pipistrellus*. Our methods are able to detect a road effect for all species, but for an accurate assessment of the magnitude of this effect on species that are less common, more data may need to be collected by increasing the number of transects at each site.

This is the first study in which road effects have been measured both during and immediately post-construction. Both stages showed significant and substantially lower bat activity and diversity in proximity to the road, with consistently large effects for all species analysed individually (*P. pipistrellus, P. pygmaeus* and *M. brandtii*/*M. mystacinus*). This suggests that the construction stage is a particularly disruptive time and that road effects occur rapidly after the commencement of construction. At our study sites large areas of land were being or had been cleared for road widening resulting in habitat loss and large, open spaces separating habitat on the two sides of the road. This highlights the need for greater consideration of the effects of the construction phase of linear transport infrastructure, and the importance of mitigating these effects prior to construction. Installing mitigation after construction has already commenced or once the road is almost complete is likely to be too late to protect local bat populations.

Changes in total bat activity were not found in proximity to one of the single carriageway roads studied (A1065). It is unlikely that this result is due to a failure of the method, given that the results on the two other single carriageway roads (A487 and A134) used the same methods and recorded a much lower number of bat passes. Abundance of *Nyctalus*/*Eptesicus* species, which may not be negatively affected by roads, was low (<1%) and therefore unlikely to have influenced the results. Individual analyses for *P. pipistrellus* and *P. pygmaeus* also did not reveal any significant or substantial changes in activity in proximity to the road. There may have been an effect on more sensitive, woodland adapted species such as *Myotis* and *Plecotus*, but more data are required to test this. Without pre-construction data and longer term monitoring we do not know whether the road has affected bats since construction, but there is no evidence to suggest that current bat populations are influenced by the road. Without further research, it is not possible to explain differences in the observed effects between roads.

Rail effects

Total bat activity, bat diversity and the activity of *P. pipistrellus, P. pygmaeus* and *M. brandtii*/*M. mystacinus* were all substantially and statistically significantly lower in proximity to the mainline in North Yorkshire. Railways can therefore have an impact on bats on a landscape scale that is comparable to that of roads. It is likely that similar mechanisms are causing these effects. Both road and rail construction result in habitat loss and degradation, and create open gaps that fragment habitat. Mortality due to collisions with trains has not been investigated. As we found with roads, not all railways appear to have a negative impact and further research is needed to understand these effects. We found no changes in bat activity or diversity in proximity to the railway in Kent, despite similarities to the rail site in North Yorkshire (e.g. size, design, train frequency), and similarly large numbers of bat passes being recorded. We cannot compare pre- and post-construction populations, but there is no measurable landscape scale effect at present. Bats may have adapted to the presence of the railway or other factors may be important. For example, the higher quality habitat surrounding the railway in Kent may be a critical factor, since high habitat quality may reduce the negative impact of roads (and therefore rail) on bats (Berthinussen & Altringham 2012b). Although habitat types along transects was very similar according to our grading system, the landscape surrounding the

railway in North Yorkshire is mainly open agricultural land, whereas in Kent there are richer bat habitats with large areas of woodland.

Time and habitat effects

Time after sunset and the habitat type at spot checks were found to have significant effects on bat activity at most sites. The habitat types found at each site are shown in Appendix H. Bat activity generally increased with habitat grade, and therefore with the height and continuity of tree and hedgerow cover along transects, a finding supported by many other studies (Russ & Montgomery 2003; Berthinussen & Altringham 2012b; Frey-Ehrenbold *et al.* 2013). Bat activity generally peaked at 90 – 100 min after sunset, with *Pipistrellus* species peaking earlier than *Myotis* species. This can be explained by the later emergence time of *Myotis* species, and may also reflect greater mobility following emergence, before bats settle to forage at their regular feeding sites. It is important that these effects follow expected patterns and are modelled alongside road and rail effects so that accurate models are produced from the data. These effects will not, however, bias the predicted road or rail effects as the GEE model takes into account the independent effects of each variable.

6.3 The effectiveness of mitigation structures

We surveyed nine mitigation structures and all yielded sufficient data for an informative assessment. We compared the proportion of bats using a structure to cross the road safely with the proportion crossing at risk of collision. Although this allows us to draw some conclusions about the current effectiveness of the mitigation, it does not tell us whether the activity of commuting bats has changed since road construction, or whether the mitigation structure has increased road permeability. In the future surveys should be conducted along bat commuting routes and potential mitigation sites before construction as well as during and for some years after. This will allow a more informative assessment of mitigation effectiveness to be made.

Setting aside the question of pre- and post-construction differences in bat activity, an important question is what proportion of bats need to be crossing the road safely for the mitigation structure to be deemed effective. This is difficult to answer because there are two important unknowns: what proportion of those at risk is killed at some point during their frequent crossings and what level of mortality is sustainable? As discussed earlier, we suggest that mitigation structures should guide at least 90% of crossing bats safely over roads to be considered effective, but local circumstance and species must be taken into account.

Over the road structures

We surveyed six over the road structures, varying from relatively simple wire bat bridges to a wide, substantial vegetated 'green' bridge (see Appendix F for details of each site). Although the sample is small, and further research is needed to draw reliable conclusions, the results suggest that the effectiveness of crossing structures increases with their size, connectivity and similarity to natural linear features.

Bat activity was very low (or absent) at all three bat wire bridges on the A11 in Norfolk. Given the number of bat mitigation structures that have been installed on this scheme we assume pre-construction activity was high to justify their construction. This suggests that there have been significant declines in bat activity since road construction began, and that mitigation is failing at maintaining local bat populations. However, without comparable pre-construction data this is unproven. Regardless of this, we can conclude that neither of the installed bat wire bridges is effective in guiding bats safely over the road. At the bridge with the highest activity, 80% of bats crossed the road at risk of collisions with traffic and only one bat of the 35 that crossed could be considered to be 'using' the gantry. None of the five bats observed at the other bat wire bridge used the wires to cross the road, and nearly all crossed at unsafe heights. These results corroborate previous research (Berthinussen & Altringham 2012a), which showed that four wire bat bridges in northern England were ineffective and used by a very small proportion of bats, with the majority crossing roads unsafely. One of the wire bridges studied by Berthinussen & Altringham (2012a) had been in situ for nine years, less than 15 m from the known commuting route. Almost all bats still used this old commuting route, suggesting that bats do not necessarily adapt to new structures with time. Note that the crossing wires on the A11 bridges form a relatively dense, v-shaped mesh, quite different to the single-wires-with-balls

previously studied (see Appendix F for photos of both). We now have evidence from six different wire bat bridges, of two designs, to show that these structures do not provide effective mitigation for bats.

Bat activity was also very low on the overbridge over the A11, which as discussed above, suggests that bat activity is likely to have declined significantly since construction, assuming this site was chosen for an overbridge based on significant pre-construction bat activity. The overbridge was also ineffective in guiding bats safely over the road. Only three bats were observed crossing near the overbridge, and none used it to cross the road. The overbridge carries a minor road over the dual carriageway below, and at the time of study had no vegetation across it and was unconnected with surrounding habitats or bat flight paths. It is unclear whether work to improve connectivity or enhance the bridge itself was planned. If such work was planned, the delay in implementing it may have had long-term effects on the suitability of the site as a bat crossing point. Previous studies have also found overbridges, built to carry minor roads or footpaths with no adaptations for bats, to be ineffective (Bach *et al.* 2004; Abbott *et al.* 2012a).

The environmental overbridge on the A487 in Wales is a unique structure, purpose built for wildlife. It differs from a conventional overbridge in that it has solid raised sides and a row of trees/shrubs in planters running across it. It does not carry traffic and is intended for use only by wildlife, but is narrower and less vegetated than a typical 'green' bridge. More bats flew over the environmental overbridge than over the road below, but 20% still crossed at risk of collision with traffic. According to our criteria this crossing structure cannot be considered to be effective at present. However, given that the majority of bats commuted across it, it has the potential to be made effective, perhaps by increasing the density and height of the vegetation. The position of the bridge maintains a pre-construction commuting route, so that bats are not required to alter their flight height or direction to fly over it, which is likely to contribute to its success as a guiding structure. Further study may help to understand why 20% of bats cross the road and how the proportion using the overbridge can be increased. It was used by most species, all showing similar patterns of use. However, use by the primary target species, *R. hipposideros*, was low. In the absence of pre-construction data we do not know if the number of bats crossing the road has remained unchanged by road construction. Additional surveys and more data would be required at this structure to determine its effectiveness for the specific target species.

Scotney Castle landscape bridge is a typical 'green' bridge. It is a relatively wide, substantial structure and although it carries a paved minor road, it is well vegetated with dense and continuous mature trees and shrubs along each side that are well connected with treelines and surrounding woodland. Although we do not know if bat activity has changed at this site since the road was originally built, we can conclude that this structure is effective in guiding bats safely over roads with 97% of bats using it to cross. At least five bat species were recorded, and foraging activity was also high over the bridge. There has been little quantitative research into the use of green bridges by bats. Ten species of bat were found to use green bridges in Germany, with higher use than conventional overbridges, but the study did not look at pre-construction data or the number of bats crossing over the road below (Bach & Muller-Steiss 2005).

Although we have only studied one green bridge the results are encouraging, and there are several factors that are likely to contribute to the effectiveness of this structure: it is well connected, providing a continuous link with bat habitat on either side of the road, the bridge follows the natural topography of the land with the road in a cutting below so that bats flying across it are not required to alter their flight height or direction, the vegetation over the bridge is mature and dense, screening noise, light and the visibility of the road below, and the minor road across the bridge carries little traffic and is closed to the public at night. Artificial lighting below the bridge may also have contributed to its success by deterring bats from crossing the road below, but further research is needed to confirm this as the effects of lighting are species-specific and can be complex.

Although green bridges have the potential to be effective crossing structures for bats over infrastructure, there are other issues that also need to be considered such as the cost, the land take required for construction of the bridge and the detrimental effects there may be on bats while it is being constructed. However, one expensive yet effective structure will always make more sense than cheaper structures that

do not work: mitigation structures must be cost effective *and* functional. Green bridges may also provide mitigation for other wildlife. Eight mammal species have been found to use Scotney Castle landscape bridge, including deer, badger and breeding dormice (National Trust, 2012), and similar structures are commonly built throughout Europe and North America for large mammals. Combining mitigation for a range of wildlife may be a cost effective solution, but would require careful planning, project management and monitoring.

A simpler over-the-road structure that may be effective on narrower roads is the 'hop-over': mature trees that overhang the road so that their crowns bridge the gap above the road (Limpens *et al.* 2005). However, this solution at present is untested and unlikely to be suitable for species that fly below the tree canopy. There may also be safety issues with trees in close proximity to and overhanging road carriageways although this is not an uncommon sight on UK roads.

Underpasses

The majority of bats at all three sites flew through the underpasses rather than over the road above. However, at both the A487 underpass and the Hopyard Farm underpass on the A465, a third of bats still crossed over the road above at risk of collision with traffic. We know that bats can be killed on roads in high numbers (Lesiński 2007; Russell *et al.* 2009; Lesiński *et al.* 2010, Medinas *et al.* 2013), and it is possible that with this proportion of bats crossing unsafely, mortality may be too high to be sustainable.

The Evesham nurseries underpass on the A465 showed high levels of use by commuting bats with just 5% crossing at risk of collision mortality on the road above. We can therefore conclude that it is effective in allowing bats to cross the road safely. This underpass is larger in both height and width than the other two underpasses studied, and also bridges a pre-existing commuting route along a tree-lined public footpath, so that commuting bats are not required to alter their flight height or direction. It has also been subject to less disturbance than Hopyard Farm underpass which was lengthened considerably during road widening, and the A487 underpass, which was constructed below a new bypass. Both Hopyard Farm underpass and the A487 underpass are also narrower and lower and do not appear to preserve original commuting routes in their entirety (even if positioned on the same alignment as original commuting routes, they may still require bats to alter their flight height to fly through them). Although these two underpasses are providing safe passage for some commuting bats (and the A487 underpass was used by all detected *R. hipposideros*, the primary target species at this site), it seems that their size and position may result in them being used less frequently by crossing bats. Similar results were found by Berthinussen & Altringham (2012a): a high underpass preserving an existing commuting route carried 96% of crossing bats below a road, but two lower underpasses not built on original commuting routes were ineffective despite attempts to divert bats to them. Underpass height was also found to be a critical factor determining the use of underpasses by bats in Ireland (Abbott *et al.* 2012a; Abbott *et al.* 2012b).

It is important to note that the Evesham nurseries underpass contained motion-activated lighting for pedestrians. This was not operational at the time of our surveys and has been permanently deactivated (because of the effects on bats, after consultation with those in charge of the footpath: Len Wyatt, pers. comm. Jan 2014), but such lighting could deter commuting bats and reduce the effectiveness of underpasses as a mitigation features.

We found up to seven species using underpasses to commute beneath roads. At all three underpasses, *Myotis* species and *R. hipposideros* flew through the underpasses in preference to crossing over the road above. *Pipistrellus* species, however, used the two smaller and narrower underpasses less frequently. At the A487 underpass, only a quarter of *P. pygmaeus* crossed through the underpass with the majority flying over the road, and *P. pipistrellus* were not recorded in the underpass at all. At Hopyard farm underpass, although we do not have species records over the road, few *Pipistrellus* flew through the underpass, which was used predominantly by *R. hipposideros*. At the larger Evesham nurseries underpass high numbers of *Pipistrellus* bats flew through the underpass with 96% of *P. pygmaeus* and 93% of *P. pipistrellus* preferring to do so rather than fly over the road above. Similar results were found in a previous study in Ireland: clutter-adapted bat species such as *R. hipposideros* and *M. nattereri* were the only species to be detected in narrow drainage pipes beneath a road, whereas all detected species flew through a larger underpass, including high numbers

of *Pipistrellus* (Abbott *et al.* 2012b). This has important implications for underpass design in relation to the species present.

7. Conclusions and best practice principles

This project confirms that roads and railways can have a negative impact on bat activity and diversity at the landscape and probably therefore the population scale. Further research is needed into the effects of rail, but bats should be given the same consideration in new railway developments as they are for roads. The methods we have developed can be applied to both, as well as other linear infrastructure.

Negative effects were found along established roads of all sizes studied (single and dual carriageway, and motorways), and also both during and immediately after road construction highlighting the importance of considering bats from the outset of such schemes and installing mitigation prior to commencing any construction work. Further research is needed into the effects of minor roads, the specific mechanisms involved in the impacts of road and rail on bats, and how these impacts affect the sustainability of local bat populations.

We conclude that the two most widespread forms of wire bat bridge do not provide effective mitigation and should not be built, particularly since there is evidence that bats do not adapt to them with time. Our results suggest that green bridges and underpasses have the greatest potential but they must be designed correctly and many factors are important such as size, position, connectivity, topography, and the density and maturity of vegetation. Such factors will be site and species-specific, and further research on specific designs is needed.

Bats need to be considered from the early planning stage of infrastructure developments, and surveys should be undertaken well in advance of decisions involving route planning, avoidance and mitigation. It is also essential that clear objectives for survey work are set, comparable pre and post-construction data are collected, and that long term monitoring of mitigation schemes is carried out to provide the evidence needed to inform future mitigation and improve on current standards. Long term monitoring is important regardless of whether an effect is found initially, as effects may be cumulative and delayed.

Standardised and objective protocols, such as those we have developed, should be used routinely and results should be documented and reported centrally. Future mitigation should be based on solutions that are proven to work, and best practice methods should be regularly reviewed and updated with the best evidence available.

We provide good practice principles for surveying bats along linear transport infrastructure based on the methods we have developed, and for mitigation based on the evidence currently available.

7.1 Best practice principles for surveying bats along linear transport infrastructure

Our survey methods are designed to provide comparable pre and post-construction data, and will provide baseline activity for the Environmental Statements that are used to assess the potential impacts of a development. They should be conducted in addition to other pre-construction surveys (using acoustic, radio-tracking and other methods) that are used in the early stages of road planning to identify bat roosts and important foraging habitats that may be destroyed or disturbed by the scheme.

We advise using a combination of landscape scale surveys to measure spatial changes in bat activity and diversity in relation to the road/railway *and* local scale surveys of specific bat flight paths to assess the effectiveness of mitigation features designed to guide bats safely under or over roads/railways. Key points for each are given below. Full, detailed instructions for survey and analysis are given in Appendices D and F.

Landscape scale effects

- Conduct the transect survey protocol (given in Appendix E) prior to construction to provide baseline data. This should be done over at least two seasons where possible.

- Repeat the transect survey protocol during and after construction. We advise surveying for a minimum of three years post-construction. Frequency of survey will depend upon the project goals but the effects of linear infrastructure on bats may persist for many years, and long term monitoring is important.

- A minimum of ten transects of 1 km length (with 10 min spot checks at 100 m intervals) must be completed per site, at the same time each year (ideally June-August). More transects may be needed if specific bat species are of interest, particularly those that are less common.

- *Nyctalus* and *Eptesicus* species should be excluded from the data for the following analyses of total bat activity, as positive impacts on these species may affect the results.

- Compare pre-construction data with during and/or post-construction data using appropriate statistical tests (such as t-tests or one way anova) to look for changes in bat activity over time. If bat activity (either total and/or for individual species) has significantly declined, the infrastructure has had a negative effect since construction.

- Analyse post-construction data using the GEE modelling method to look for changes in total bat activity in proximity to the infrastructure, and repeat the analyses for all species with sufficient data. The analysis may also be repeated for bat diversity.

- If the effect of distance from the infrastructure is positive (with a significance level of $P < 0.05$) or bat activity (either total and/or for individual species) is predicted to increase by at least 20% (regardless of statistical significance) between 0 and 1 km from the infrastructure, the effect is considered to be detrimental to local bat populations. Rare or vulnerable populations may need special consideration.

- Results of statistical tests and models should be presented in reports in text and table format, and data should be displayed visually using boxplots.

Local scale & mitigation

- Identify bat commuting routes that will be severed by the scheme prior to construction (as detailed in Appendix G) to inform the placement of mitigation structures.

- Before construction, complete a minimum of six 60 min dusk or dawn surveys at each location where mitigation is to be installed (following the local scale survey protocol in Appendix G). Longer surveys, running later into the night, may be necessary if vulnerable, woodland-adapted species are involved.

- Repeat the local scale survey protocol at each location where mitigation is installed at the same time each year during construction and post-construction. We advise surveying for a minimum of three years post-construction (frequency of survey will depend upon the project goals but the effects of linear infrastructure on bats may persist for many years, and long term monitoring is important).

- For schemes that have already been completed, surveys may be carried out post-construction only to provide a basic assessment of the mitigation. However, this will not give a complete picture of effectiveness or the impact of the scheme and is insufficient for future schemes.

- Compare the total number of bats crossing at each site before, during and after construction, and the number of bats considered to be using the mitigation structure in question (according to set definition of 'use'), and the number crossing the scheme at risk of collision with traffic.

- Mitigation structures are considered effective when the number of bats using the commuting route has not declined substantially (by a statistically significant decline of 10% or more) since construction, *and* at least 90% of crossing bats are using the structure to cross safely. Special consideration will need to be given to rare and vulnerable populations. Not all species may be affected in the same way.

- Results should be presented in reports in text and table format, and data should be displayed visually using boxplots. An optional kernel density estimation plot may also be produced to show the position and density of crossing bats at a site.

The results from both methods above should be considered, to give an overall picture of the impact of the infrastructure scheme on a landscape and local scale, and the effectiveness of any mitigation structures over the scheme.

The aim is for linear transport infrastructure schemes not to have a detrimental impact on local bat populations, and for effective mitigation of any negative impacts. This means that bat abundance in the landscape surrounding the scheme should not drop significantly during or after construction, the number of bats using a commuting route should not decline substantially during or post-construction, and at least 90% of bats crossing the scheme should do so safely without risk of traffic collision. It is also important to look at individual species, particularly those at risk due to rarity, ecology or proximity to the development.

7.2 Best practice principles for bat mitigation along linear transport infrastructure

Sample sizes for most mitigation structures are currently small, and further research is needed to make specific recommendations, especially on structure design. Based on the evidence available at present, we suggest that the best solution is a combination of underpasses and green bridges along the length of linear infrastructure to maintain bat commuting routes and increase the permeability of the infrastructure. The design and placement of mitigation structures will be site and species-specific. However, the following general principles are likely to contribute significantly to the success of these crossing structures:

- ***Mitigation should be integrated into the scheme from the earliest opportunity***
  Mitigation should be considered during the planning and design stage of the infrastructure so that it can be incorporated effectively.

- ***Crossing structures should be placed on the exact location of existing bat commuting routes***
  Attempts should not be made to divert bats from their existing commuting routes.

- ***Crossing structures should not require bats to alter flight height or direction***
  This will depend on the topography of the site. If the road is to be elevated above ground level an underpass may be used to preserve the commuting route below it, or if the road is in a cutting a green bridge may be used to carry the commuting route over the road.

- ***Crossing structures should maintain connectivity with existing bat commuting routes*** Connectivity must be maintained with undisturbed bat flight paths (e.g. treelines, hedgerows, woodland rides and streams), and bat habitat (e.g. woodland) within the surrounding landscape. Crossing structures should not be exposed or sited within open ground.

- ***Over-the-road structures such as green bridges should be planted with vegetation***
  Vegetation should be continuous and connected (see above) and sufficiently mature before road construction (e.g. by planting either relatively mature trees or fast growing tree species in advance of construction commencing).

- ***Underpasses should be of sufficient height***
  Underpasses should be as spacious as possible with height being the critical factor. The minimum requirements for underpass height will be species-specific. Required heights will generally be lower for woodland-adapted species (~3 m) compared to generalist edge-adapted species (~6 m), but larger underpasses will accommodate more species

- ***Green bridges should be of sufficient width***
  In addition to being vegetated, green bridges should be as wide as possible, to provide a large area for bats to commute across. Further research is needed to determine exact dimensions. We found a 30 m wide green bridge to be effective in this study.

- ***Crossing structures should be unlit***
  The effects of light on bats are species-specific and lighting should be avoided.

- ***Access and connectivity must be maintained***
  It is important that access to crossing structures is maintained (e.g. grilles should not be installed on underpasses) and that connecting vegetation is retained indefinitely or for as long as the mitigation structure is required.

- ***Disturbance should be minimised during installation of mitigation structures***
  For example, by limiting noise and light pollution along the bat flight path, minimising vegetation clearance, installing suitable temporary crossing structures (which should also be subject to

monitoring and evaluation), completing the installation as quickly as possible and ideally avoiding the summer months when bats are most active.

This advice is based on the evidence currently available. The aim is that with further research and appropriate monitoring and documenting of results from future mitigation schemes, the evidence base can be improved. This advice should be reviewed and updated regularly to improve on current standards.



Timings and Longevity Predictions for Minsmere

Non-technical Summary

August 2006

BLACK & VEATCH
building a world of difference™

TIMINGS AND LONGEVITY PREDICTIONS FOR MINSMERE NON-TECHNICAL SUMMARY

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TIMINGS AND LONGEVITY PREDICTIONS FOR MINSMERE NON-TECHNICAL SUMMARY

1. INTRODUCTION

Black & Veatch (B&V) were appointed in 2001 by the Environment Agency to address concerns over the long-term viability of the sea defence system at Minsmere on the Suffolk Coast. The section of coastline under study extends from Dunwich Cliffs in the north to the flood defence embankments protecting Sizewell power station to the south.

The current front line flood defence along this frontage consists of a dune embankment. Seaward of the embankment lies a relatively narrow sand & shingle beach that slopes to the waterline. A tidal sluice inland of the dune embankment approximately mid-way along the frontage drains the 82km^{2}catchment. The area of land drained by the sluice includes the Minsmere Nature Reserve of the Royal Society for the Protection of Birds (RSPB).

This report aims to provide a non-technical summary of the Timings and Longevity Predictions for Minsmere Report (Black & Veatch, 2006) to promote understanding of the way forward. This report summarises the current standard of protection and residual life of the existing defences along the Minsmere frontage in order to determine whether works are required and if so, when they should be undertaken. In addition, in order to design and cost any works required to provide specific standards of defence, it is necessary to determine what crest levels are required to achieve a specified standard of defence. For further information regarding the issues raised in this report, the reader is referred to the report that this is a summary of and any reports referenced therein.

2. STANDARD OF DEFENCE AND RESIDUAL LIFE

2.1 Standard of defence assessment methodology

The primary defence along the Minsmere frontage consists of a mixed shingle and sand sloped beach backed by a relic dune system. There is evidence that the upper section of the landward side of this front dune system has been built up by man with made ground consisting of slightly clayey sandy gravel. At the northern end of the frontage the dune system is low and fronted by a narrow stretch of beach. Further south the dunes widen and the fronting beach becomes considerably wider.

A secondary defence exists along the Minsmere frontage from just south of Minsmere cliffs to about 100m north of the sluice.

An examination of the current standard of defence for four defence scenarios was undertaken:

- Scenario 1 - The primary defence along the entire frontage in isolation;
- Scenario 2 - The secondary defence along the northern frontage assuming that the primary defence has been removed;
- Scenario 3 - The secondary defence assuming a small breach in the primary defence;
- Scenario 4 - The secondary defence along the northern frontage assuming that the existing front crest (primary defence) has moved back under storm events building the beach up against the secondary defence to create a berm and thereby providing additional protection.

Three different overtopping methodologies were employed to estimate the standard of defence for the existing Minsmere primary defence and secondary defence banks at each topographic survey (1 to 20) and EA beach survey (S1B2 to S1B4) location along the Minsmere frontage. The location of each of these survey point is shown in Figure 1.

As yet, English Nature have not defined a necessary standard of protection for maintenance of the site features. A range of standards from 1 in 10 to 1 in 50 is being examined for costing purposes.

2.2 Residual life assessment methodology

Study of coastal processes at Minsmere (reported in Coastal Processes Report, Black & Veatch, 2005) showed evidence of recent erosion along sections of the frontage. If current erosion rates continue there will be a time when the current dune embankment which forms the primary defence will become threatened and failure of the defence is likely to occur resulting in flooding of the hinterland.

The time of failure was assumed to be when the most landward extent of the bank/dune providing a 4mAOD crest level (approximately a 1 in 10 standard of defence) became threatened by erosion. An examination of the residual life based on erosion rates was undertaken for three of the defence scenarios above as follows:

- Scenario 1 - The primary defence along the entire frontage in isolation;
- Scenario 2 - The secondary defence along the northern frontage assuming that the primary defence has been removed; and
- Scenario 4 - The secondary defence along the northern frontage assuming that the existing front crest (primary defence) has moved back under storm events building the beach up against the secondary defence to create a berm and thereby providing additional protection.

For each beach survey point, a worst case, best case and most likely rate of change of beach level were determined based on assessments undertaken in the Coastal Process Report (Black & Veatch, 2005). This information is based on EA survey data (1992 to 2005) and Admiralty Charts (1976 to 2005). The additional increase in the rate of erosion due to sea level rise was also included. An estimate of the time before the defence is likely to be compromised by erosion processes was undertaken for the three scenarios for each erosion case.

Due to the methodology adopted, scenarios two and four yield the same result as the additional material in place under scenario four is not given special consideration. Therefore only one residual life estimate was made for each of the primary and secondary banks.

2.3 Results

Table 1 shows the predicted standards of defence and residual life estimates for the primary and secondary defence for each profile along the Minsmere frontage. The predicted standards of defence in relation to the position on the Minsmere frontage are shown in Figure 2. Figure 3 shows the predicted residual life along the Minsmere frontage. The standard of defence (SOD) results for scenario 2 are not shown as this was felt to be too much of a pessimistic approach.

In terms of standards of defence, from Table 1 the following conclusions can be drawn:

- The primary defence from profile 8 southwards require no works at the current time as the existing standard of defence is 1 in 50 or above and this is felt to be sufficient.
- North of the profile 8 the standard of defence of the primary defence drops to as low as 1 in 2 to 1 in 10 in some places. This defence line has been overtopped in recent years confirming the low crest level at this point.
- For the secondary defence, under the current scenario (scenario three), as long as the current primary defence is still in place, then if water gets to the secondary defence it still provides protection to a minimum 1 in 20 standard.
- For the secondary defence, if the current dune ridge rolls back to protect the current secondary defence line, as is thought most likely (scenario four), then without work the secondary defence would provide a standard of protection of between 1 in 2 and 1 in 100. The southern extent of the secondary bank (profiles 7 to 10) has considerably lower crest levels, resulting in a standard of defence as low as 1 in 2.

In terms of residual life, from Table 1 the following conclusions can be drawn:

- The primary defence bank on the northern frontage may breach within the next five years, the most vulnerable area being at the location of profiles 3 and 4. This is where the beach seaward of the bank is currently at its narrowest and there have been actual incidences of overtopping at this location.
- The entire most northern section of the primary defence (profiles 1 to 5) will be lost within 30 years – flood protection will have to rely on the secondary bank alone.
- The entire northern half of the secondary bank (profiles 1 to 5) may be lost within 50 years (with breaching commencing within about 20 years time), indicating that realignment may be required at this location in about 20 years time.
- On the southern section of the frontage (i.e. south of the sluice), the analysis highlights the vulnerability of the primary defence at the location of profiles S1B4 and 18 as the defence embankment mat be breached within 20 years. A localised blister bank could be constructed behind these defences in the medium term to ensure defences are maintained.

	Primary Defence		Secondary defence		
Profile	SOD	Residual life against erosion	SOD until primary defence eroded (scenario 3)	SOD assuming primary defence creates a berm (scenario 4)	Residual life against erosion
Prof 1	200	30	+500	100	48
Prof 2	10	22	+500	50	42
Prof 3	2 to 10	1	+500	20	29
Prof 4	10	5	+500	20	22
Prof 5	10	23	+500	100	47
Prof 6	2 to 10	245	+500	50	569
S1B2	20	177	50	20	399
Prof 7	5	239	20	2	239
Prof 8	50	1496	20	2	1496
Prof 9	50	1148	+500	2	1148
Prof 10	200	944	+500	5	944
Prof 11	500	1840			
S1B3	200	No Failure			
Prof 12	500	No Failure			
Prof 13	500	136			
Prof 14	500	144			
Prof 15	500	139			
Prof 16	500	153			
Prof 17	200	159			
S1B4	200	19			
Prof 18	200	11			
Prof 19	200	107			
Prof 20	50	125			

Table 1. Summary of standard of defence and residual life for Minsmere frontage

2.4 Loss of SAC habitat

Loss of SAC vegetated shingle habitat has notably occurred along the north of the Minsmere (profiles 1 to 10). Table 2 shows the current width of beach between the MHWS line and the secondary defence, and an estimate of the predicted beach width after 20 and 40 years. The reduction as a percentage is also shown.

The results in Table 2 show that the SAC habitat will be dramatically reduced over the northern section (profiles 1 to 5) as a result of coastal squeeze. However, any realignment of the frontage will cause loss of the SPA habitat behind. English Nature have advised that the SPA habitat should be maintained for 29 years to allow compensation habitat to be established for any planned realignments

in the future. In 20 years time, the defence line over profiles 1 to 5 should be realigned to provide a minimum 45m width of beach seaward of the defence throughout the design life of the realigned defence in order to allow SAC habitat to migrate landward as sea levels rise.

Profile	Distance between MHWS and the secondary defence			% reduction in distance	
	Current	Estimated in 20yrs	Estimated in 40yrs	Estimated in 20yrs	Estimated in 40yrs
Prof 1	65	52	37	20	43
Prof 2	61	46	29	25	52
Prof 3	50	35	18	30	64
Prof 4	44	28	16	36	64
Prof 5	62	48	33	23	47
Prof 6	79	78	75	1	5
Prof 7	87	86	84	2	3
Prof 8	100	100	97	0	3
Prof 9	98	98	98	0	0
Prof 10	98	98	98	0	0

Table 2 - Predicted reduction in distance from MHWS to secondary defence

3. IDENTIFIED WORKS

From the assessment of standard of defence and residual life, it has been possible to identify sections of the current embankment defence system that require works in the short and medium term, and to estimate the timings of these works.

It should be noted that the timings are based on erosion rates which are highly variable. These timings are suggested only based on the current available information. Monitoring of the frontage should continue and these timings adjusted as erosion rates change.

Works required to the sluice to resolve deterioration issues have also been summarised here.

3.1 Short term works

The following works have been identified to be required to be undertaken in the short term (0 to 20 years):

- Undertake strengthening and raising works to the secondary defence from Dunwich Cliffs to profile 5 to an agreed standard of defence. Any works should be designed to last 20 years, at which point the embankment will require realignment (see medium term works). The delay in realigning will enable compensatory SPA habitat to be established prior to it being lost.
- Undertake strengthening and raising works to the secondary defence from profile 5 to the sluice to an agreed standard of protection. Any works should be designed to last 50 years, at which point the embankment will require realignment (see long term works).
- Identify and secure compensatory habitat for short and medium term realignment works to allow time to establish.
- Replacement of the concrete pipes at the toe of the outfall to avoid undermining or collapse of the outfall in a severe storm.
- Construction of a blister bank behind the existing defence over profiles 17 to 19 in 15 to 20 years time. Although the results indicate that this work will be required earlier, an assessment has shown that the crest level of the eroded bank will be sufficient to provide a 1 in 10 standard of protection in 20 years time. Works may be required earlier if a higher standard of protection is required.

3.2 Medium term works

The following works have been identified to be required to be undertaken in the medium term (20 - 50 years):

- Realign the defence line at profiles 1 to 5 in 20 years time by constructing a new earth embankment from Minsmere Cliffs to the existing cross bank. The existing cross bank will also require strengthening and raising to the agreed standard of protection.
- Identify and secure compensatory habitat for long term realignment works to allow time to establish.
- The condition survey of the sluice identified that the outfall pipes will require replacement in 30 – 50 years time. However, this work should be undertaken in consideration with the future location of any new sluice constructed in the longer term.

3.3 Long term works

In the long term it is recognised that a more substantial and sustainable realignment of the coastal frontage will be required. At present it is estimated that this will be required in 50 years time. In the meantime, monitoring of the coastline will continue. Realignment of the defences will require the relocation of Minsmere sluice.

4. REQUIRED CREST LEVELS

The overtopping methodology used previously was used to estimate the required crest level height for a 1 in 10, 20 and 50 standard of defence earth embankment, assuming that the rate of overtopping discharge should be maintained at less than 5 l/s/m run. Sea level rise was included for the life time of the defence. Additions were also made to allow for uncertainty and settlement. The required crest levels for the defence embankments are shown in Table 3.

	Standard of Defence Provided		
	1 in 10	1 in 20	1 in 50
Secondary bank profiles 1 - 5	3.9	4.2	4.4
Secondary bank profiles 6 - 10	4.5	4.9	5.3
Realigned bank profiles 1- 5	3.5 – 4.5	3.7 – 4.9	3.9 – 5.3
Blister bank profiles 17 - 19	3.5	3.7	3.9

Table 3 – Required crest levels for identified works

Required crest levels for the southern section of the secondary bank (profiles 6 – 10) are greater than that for the northern section (profiles 1 – 5), as the southern section will last 50 years (and hence includes for sea level rise of 50 years), whereas the northern section will be replaced by the realigned bank in year 20.

For this assessment it has been assumed that the realigned bank will be located just seaward of the current drainage line that crosses Coney Hill, thereby allowing the drain to be converted to a soak dyke, reducing cost. This realigned defence will join the existing cross bank, where it will tie into the existing secondary bank. A range is given for the required crest levels as if it is assumed that no wave attack will impact this defence, the crest level may be estimated by considering the surge water level only (plus the additions for sea level rise, uncertainty and settlement).

The blister bank behind the existing dune system over profiles 17 to 19 will not be exposed to wave attack. Therefore the crest levels only take into account surge water level only (plus additions for sea level rise, uncertainly and settlement).

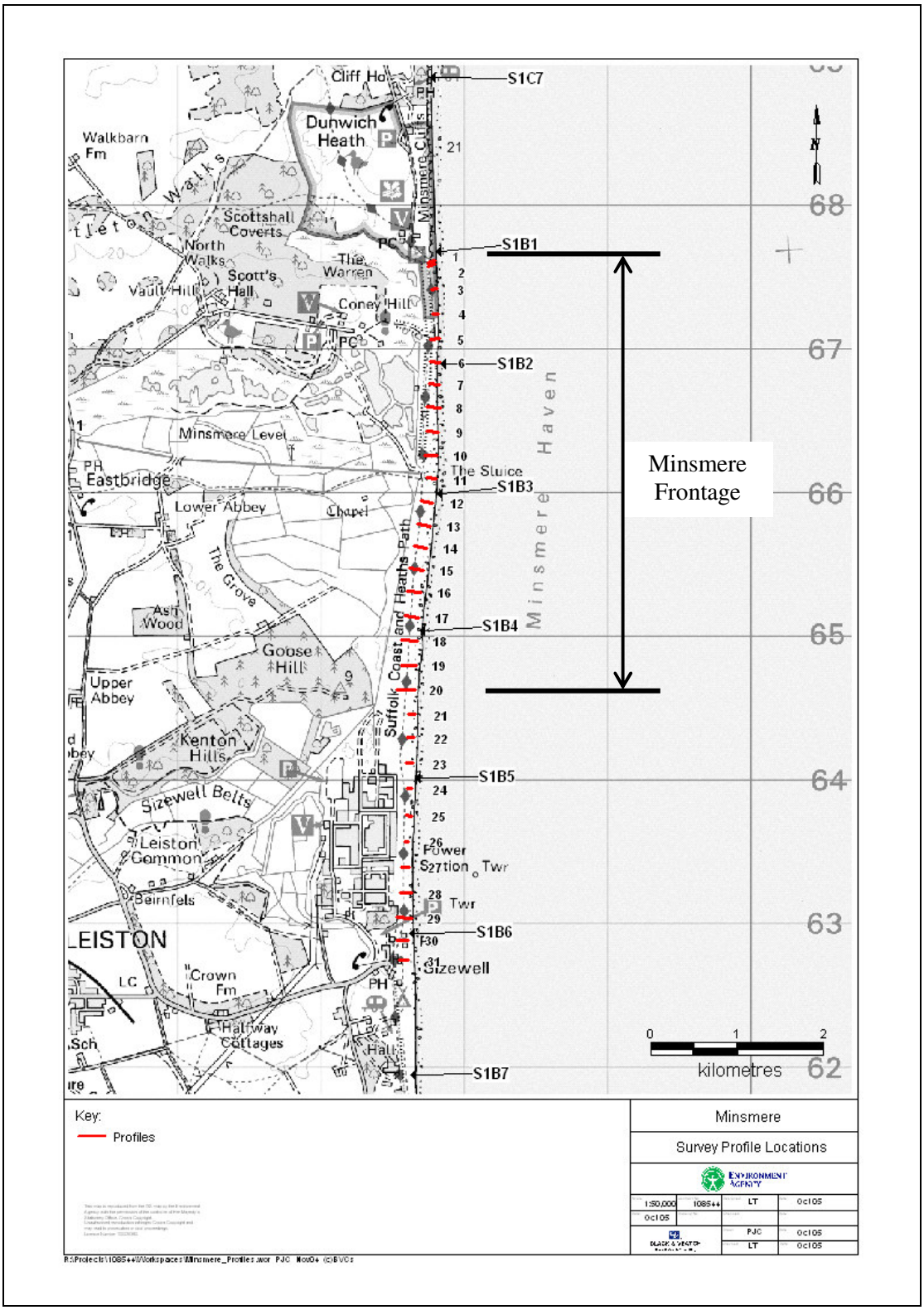


Figure 1 - Location of EA and topographic profiles

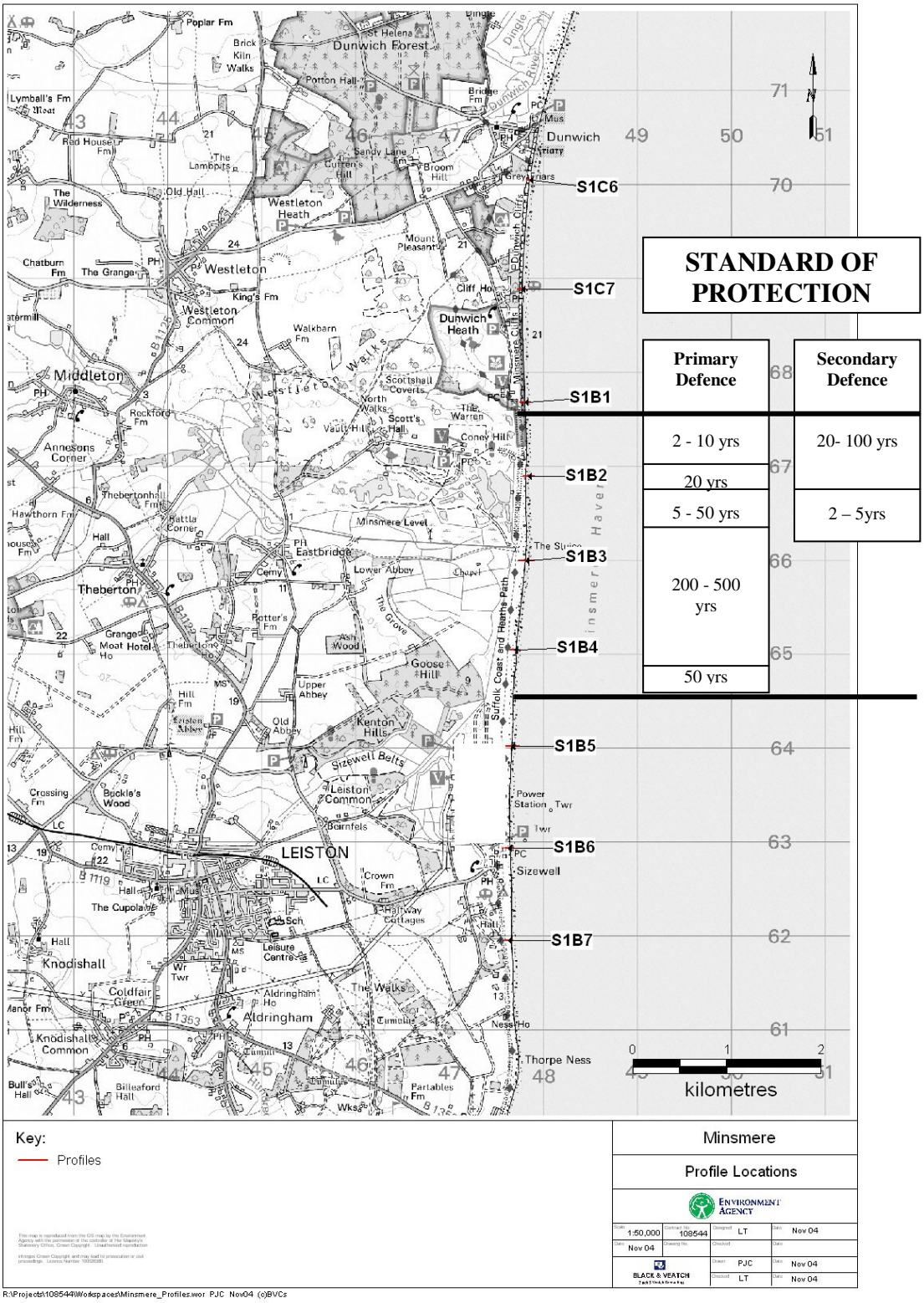


Figure 2 - Standard of protection along Minsmere frontage

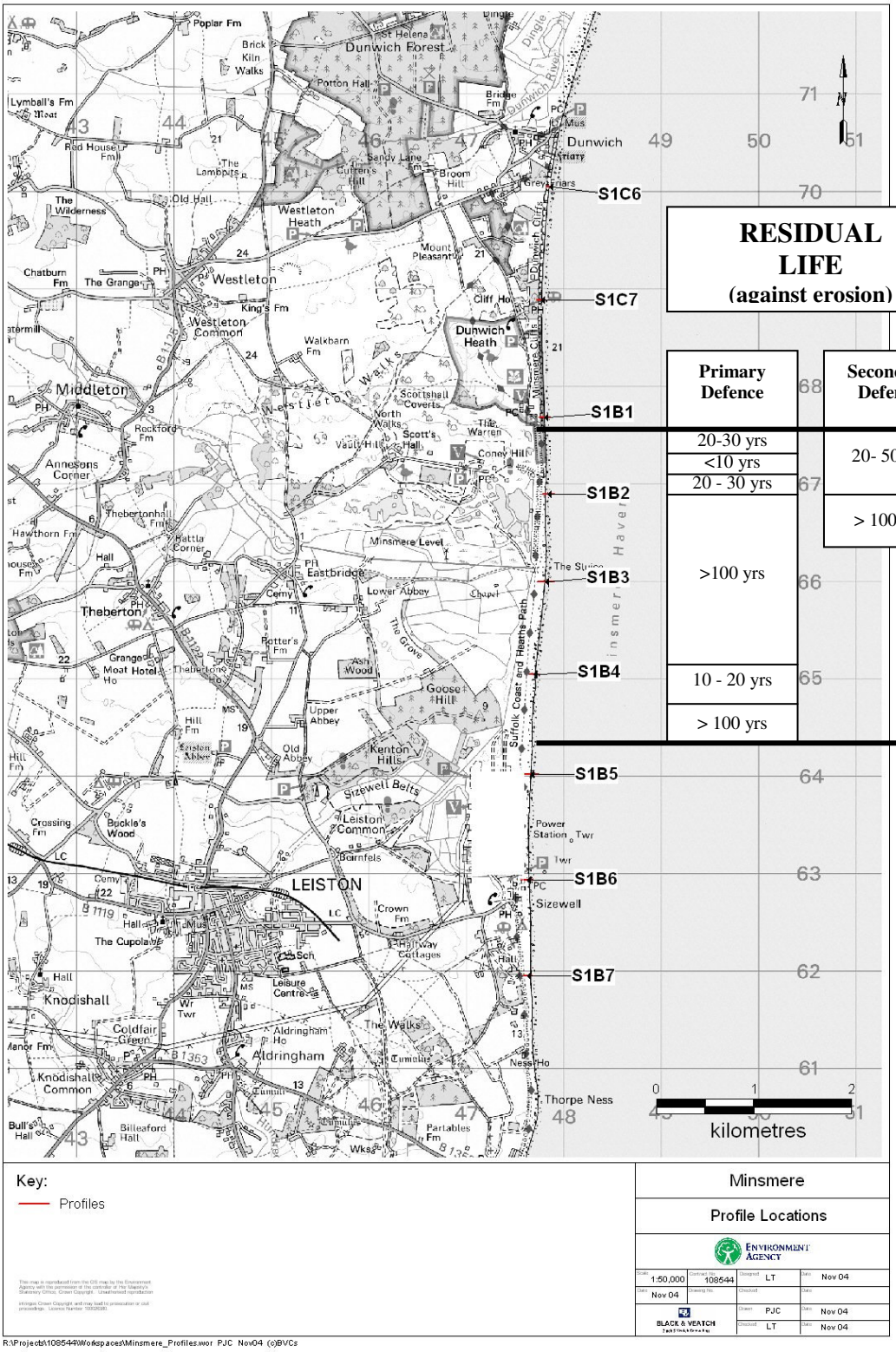


Figure 3 - Predicted residual life along Minsmere frontage

Avocetta N° 18: 9-11 (1994)

Prey selection by parents and chicks of the Little Tern *Sterna albifrons*.

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Abstract - The diet of Little Tern chicks was studied in order to analyze the prey choice of a species that carries single, unbroken prey to its nest. Chicks less than 5 days old were fed with thinner fish than chicks older than 10 days. The average prey size was smaller for younger chicks. Prey left uneaten by chicks included larger, bulkier fish, shrimps and potentially poisonous fish. The parents select prey according to the chicks' ability to swallow, and a further selection is accomplished by the chicks themselves.

Introduction

Most studies on diet and prey choice of central place foragers deal mainly with multiple prey loaders and, to a lesser extent, with single prey loaders which are able to break-up large prey items in order to feed themselves or their chicks (Houston and McNamara 1985). Other species are single prey loaders which do not break the prey; their chicks swallow the prey whole. This constraint is particularly important, because the ability of the chicks to swallow items of increasing size continues throughout their growth, and the adults' optimal prey choice changes accordingly.
In this paper we analyze the feeding behaviour of the Little Tern *Sterna albifrons*, as a model for studying diet and prey choice of single prey loaders that do not break food items. Previous studies described the diet of the Little Tern (Glutz Von Blotzheim and Bauer 1982, Cramp 1985, Bogliani *et al.* 1992, Dementev *et al.* 1966), its foraging niche (Isenmann 1979, Dubois 1982, Fasola *et al.* 1989), and chick behaviour (Davies 1981). However no attention has been paid to optimal prey selection performed by the adults in relation to the age of chicks.

Methods

Data were collected in the Comacchio lagoon (North Adriatic coast), a very old 100 km² lagoon, presently banked up at the edges and managed as fish pond. The Little Terns breed on islets in the middle of the lagoon and search for food mainly within 4 km of the colony (Fasola and Bogliani 1990). Nests are placed mainly on bare ground and beaches covered by empty *Cardium* shells and with patches of halophytic vegetation.
Observations were made from a hide close to 40 nests from 22 June to 5 July 1983. Prey items carried by adults were identified to the lowest possible taxon, and the total length was estimated using bill length as a reference. No adjustment was made for possible systematic bias in the length estimate (Goss-Custard *et al.* 1987). One set of observations was limited to two age classes: chicks younger than five days (179 prey items) and chicks older than 10 days (107 items). Both age groups were observed simultaneously within the same colony, therefore any effect due to the uneven availability of the different prey can be excluded. It was impossible to know the sample size of chicks, because they were usually hidden among vegetation and were observed only when adults arrived with food. For the same reason it was difficult to have an accurate measurement of the handling time; only in few instances it was possible to observe chicks while disgorging food items.
In order to compare abandoned with offered prey,on 2 July 1983 we removed all prey items abandoned near the nests; from 2 July-5 July we carefully searched for recently abandoned prey items twice a day.

Results

Around the nests, 164 abandoned prey were collected; over the same four days, 784 feedings were observed.

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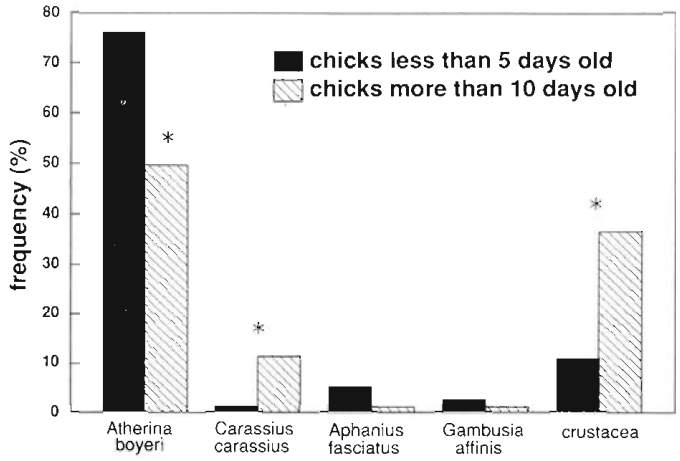


Figure 1 - Relative frequency of different prey fed to chicks aged less than 5 days (n=179) and more than 10 days (n=107). The asterisks indicate a significant difference (X^2 test, d.f.=1, p<0,01).

Table 1 - Length of the main prey delivered to chicks of differrent ages (mm).

	age of chicks							
	less than 5 days			more than 10 days			t	p
	mean	sd	n	mean	sd	n		
Atherina boyeri	34.6	6.9	136	47.7	9.5	53	9.3	<0.001
crustacea	40.1	5.7	19	51.4	2.7	39	10.3	<0.001

The frequency of occurrence of different prey items varied with the age of chicks (Figure 1). Young chicks received more Sand Smelts *Atherina boyeri* than older ones (X^2=19.7, d.f.=1, P<0.01), while the opposite occurred for the Crucian Carp *Carassius carassius* (X^2 =12.6, d.f.=1, P<0.01). Chicks over 10 days were fed with more crustaceans than were chicks less than five days old (X^2=26.1, d.f.=1, P<0.01). Young chicks received 27% smaller Sand Smelts and 22% smaller Crustacea on average (Table 1).

Overall, chicks less than 5 days old received prey items which being short and narrow were easy to swallow.

Crustaceans are probably difficult for small chicks to swallow as a consequence of the numerous appendages protruding from their bodies.

Abandoned prey was not a random sample of the offered prey. Relatively few thin fish such as Sand Smelt and Mosquito fish *Gambusia affinis* were abandoned; more bulky fish such as the Crucian Carp and Gobiidae and crustacea were overrepresented (Figure 2). Abandoned fish items were larger, on average, than offered ones (Table 2). Very few attempts at prey disgorging were observed, since chicks often took shelter immediately after receiving prey from the adult and went out of view.

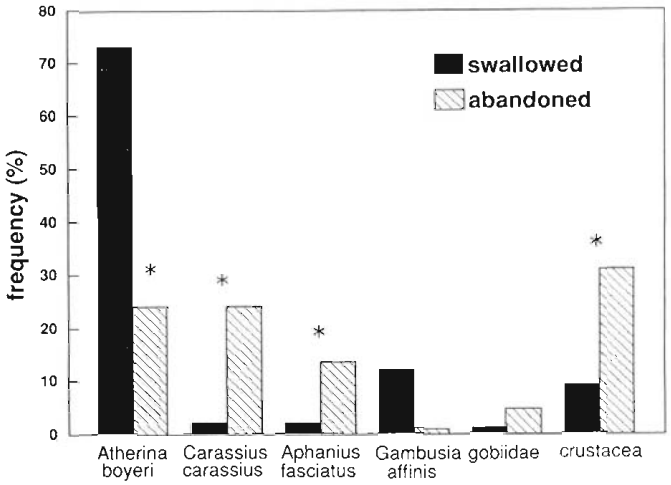


Figure 2 - Relative frequency of different prey swallowed by chicks (n=784) and abandoned uneaten near the nest (n=164). The asterisks indicate a significant difference (X^2 test, d.f.=1, p<0,001).

Table 2 - Length of the main prey observed and abandoned near the nests (mm).

	offered items			abandoned items				
	mean	sd	n	mean	sd	n	t	p
Atherina boyeri	34.7	20	574	39.9	10.4	40	3.1	<0.01
C.carassius	32.1	4.0	21	43.8	10.4	40	10.3	<0.001
Aphanius fasciatus	26.6	8.5	16	39.5	7.5	21	4.8	<0.001

Discussion

Adults selected prey of various length in relation to their chick's ability to swallow them, older chicks receiving larger prey. Obviously, a large chick is able to swallow a small prey, but this prey would not be an optimal choice for the parents. They may be qualitative differences between the pairs with young and old chicks at the same time, the latter being early layers and probably therefore older, more experienced birds. A change in the length of prey delivered to chicks of varying age has been observed in some

other species, such as the Kingfisher *Alcedo atthis* (Bogliani and Massara 1990), the Sandwich Tern *Sterna sandvicensis* (Veen 1977) and the male Common Tern *Sterna hirundo* (Wiggins and Morris 1987). In the latter species the female did not increase the length of prey as chicks got older. In the Grey Heron *Ardea cinerea*, the maximum size of fish swallowed by chicks increases with chick age (Moser 1986). Furthermore, heron chicks less than 20 days old were fed more frequently with small prey and less frequently with bulky fish than were older chicks, as was also observed with the Little Tern of varying ages in the Comacchio Lagoon. However it is not clear if the Little Tern selects for prey size and prey type according to the age of chicks, or if it simply selects for prey size; the difference in prey species could be due to the fact that small prey include species which are different from larger prey. Few items were not eaten by chicks and these tended to be those which were either more difficult to swallow than average ones, because of their shape. (e.g. fat fish), sometimes with numerous appendages causing handling problems, (e.g. shrimps), or possibly because of their bitter taste, (e.g. *Aphanius fasciatus*), which is poisonous to small mammals (Cavicchioli 1962).

There is no evidence that selection against bulky prey was due to factors other than bulk, but in the crustaceans the proportion of indigestible material they contain may reduce their palatability. This non-optimal foraging by adults would probably warrant investigation. However it seems that the amount of abandoned prey is a small fraction of prey loads brought to the nest by parents.

The difference between abandoned and swallowed prey cautions against the analysis of abandoned prey to estimate the diet of terns, as attempted by some authors (Boldreghini *et al.* 1988, Atwood and Kelly 1984). Abandoned prey are likely to be unwanted by chicks.

Acknowledgements - Thanks are due to SIVALCO, which allowed access to the Valli di Comacchio, to Alberto Meriggi for his criticisms of a draft of this paper and to Robert Coates for checking the English.

Riassunto - I pulcini di Fraticello delle Valli di Comacchio vengono imbeccati dai genitori con pesci piccoli e di forma affusolata quando hanno meno di 5 giorni; in seguito ricevono in maggior misura prede piu grandi, piu tozze o con appendici. Una parte delle prede portate dagli adulti resta abbandonata a terra. Questa frazione comprende pesci più grandi e di forma più tozza della media, crostacei con lunghe appendici e pesci potenzialmente velenosi. Gli adulti selezionano le prede in relazione all'abilita dei pulcini di ingoiarle; una ulteriore selezione viene operata probabilmente dai pulcini stessi, che scartano quelle meno idonee.

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NESTLING DIET AND PARENTAL PROVISIONING BEHAVIOUR IN THE MARSH HARRIER (*CIRCUS AERUGINOSUS*)

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Abstract. Three Marsh Harrier's (*Circus aeruginosus*) nests were monitored during the breeding season at two lakes in the Mazurian Lakeland of NE Poland. Female Harriers visited the nests more often than males and stayed with their nestlings much longer than males. At all nesting sites, the main prey delivered to young Marsh Harriers were *Microtus* voles (56% of all prey items) and other rodents (12.5%). However, the composition of the Marsh Harrier's diet differed significantly among the three nests. Waterfowl and the domestic chicken (*Gallus domesticus*) constituted 71% of the biomass consumed. No significant differences were recorded between prey items delivered to the nests by males and females. However, some prey, such as the Crested Grebe (*Podiceps cristatus*), Coot (*Fulica atra*) and young brown hares (*Lepus europaeus*), were brought to the nests only by females. The overall prey biomass delivered to nestlings by females was approximately five times larger than that delivered by males. The diet composition determined from carcass analysis and via observations differed significantly from the data obtained from pellet analysis.
Key words: diet variation, pellets, provisioning behaviour, Marsh Harrier, Poland

Introduction

In recent decades, numerous studies of the Harriers' (*Circus* sp.) biology have been undertaken in Europe and North America. Most of them concern breeding behaviour (Simmons *et al*. 1986; Barnard *et al*. 1987; Millon *et al*. 2002; Wiącek 2006a), mating systems (Amar *et al*. 2005), territorial protection (Temeles 1989a; Wiącek 2006b), hunting activity (Schipper *et al*. 1975; Schipper 1977; Temeles 1989b; Kitowski 2007), food niches (Underhill-Day 1989; Clarke *et al*. 1993; Redpath *et al*. 2002; Garcia & Arroyo 2005) and the impact on prey populations (Underhill-Day 1989). In Poland, studies by Witkowski (1989) and Buczek and Keller (1995) have considerably extended knowledge of the Marsh Harrier's (*Circus aeruginosus*) breeding ecology. This paper presents data on the Marsh Harrier's diet composition and provisioning behaviour observed at three nesting sites within lake habitats during the breeding season. Moreover, we compared different methods used for the investigation of the Marsh Harrier's diet. This topic was dealt with in several studies which revealed the necessity for the combined use of direct (observations, images) and indirect (analysis of prey remains, pellets) investigation methods (Simmons *et al*. 1991; Redpath *et al*. 2001; Selås *et al*. 2007; Tornberg & Reif 2007). This study compared data pooled from the analysis of collected prey remains and visual observations to those obtained from pellet analysis, and estimated differences between them, e.g. related to the load-size effect (Sonerud 1992).

Material and methods

Three Marsh Harrier's nests with nestlings were observed from the beginning of June until mid-July 1997. Two nests were located at Lake Inulec and one at Lake Śniardwy in the Mazurian Lakeland, in north-eastern Poland (53°40'N, 21°30'E). The nests were built in reeds (*Phragmites australis*), within the lake's littoral zone. The nesting site at Lake Śniardwy was surrounded by broad reed beds, and located at a distance of about 50 m from the shore. At Lake Inulec, the nests were located on narrow reed strips, at about 5 m and 8 m from the shore, respectively. At all the sites, the terrestrial habitats were similar, consisting of belts of alder-woods overgrowing lake banks adjacent to vast pastures and meadows. The important difference between the nesting sites was the presence of a small village in the neighbourhood of two nests at Lake Inulec. At the Lake Śniardwy site, the nearest settlements were located over 1 km away. Another important difference concerning the aquatic habitat was the presence of a large waterfowl colony

(dominated by the Crested Grebe (*Podiceps cristatus*) and the Coot (*Fulica atra*)) at Lake Śniardwy, located about 300 m from the Marsh Harriers' nest. At Lake Inulec, waterfowl nests were also present but scattered along the shore.

The Marsh Harrier's nests were monitored from shelters set at a distance of about 10 m from each nest. The behaviour of birds outside the nests was not observed. Monitoring of the nests was carried out at all possible hours of the day and its overall duration was 208 hours. The duration of a particular observation session was 2–4 hours. Binoculars 10 × 50 were used for observations. The nest at Lake Śniardwy (Nest 1) was observed for 94 hours and the nests at Lake Inulec for 59.5 hours (Nest 2) and 54.5 hours (Nest 3) respectively. In all the nests, chicks clutched within the last ten days of May. Observations of the nests were started 7–10 days after the clutching of chicks. There were 4 nestlings both in Nest 1 and in Nest 3, and 2 nestlings in Nest 2. Observations were not noticed to exert any negative impact on the Harriers' behaviour and nesting success and all chicks fledged successfully. During the observation, all arrivals made by parents, as well as the time spent by males and females at the nests, were recorded. Prey items delivered by adults were identified, if identification was possible. During observations it was not possible to record aerial food transfers between mates, so it is hard to estimate what proportion of total food transfers from males to nestlings was mediated through females. At the end of each observation session, food remains and pellets were collected from the nest, including prey remains delivered beyond observation sessions. Remains found in the nest, if different from the observed prey delivered by parents, were treated together with food items whose delivery was monitored. Pellets were oven-dried at 50°C. All undigested remains were identified according to the key provided by Pucek (1981) and assessed in relation to the comparative content of bones, scales and feathers. Altogether, 183 prey remains and instances of prey delivered by parents were recorded; 174 prey items were found in 113 pellets. The number of prey items in pellets was calculated according to standard methods (Raczyński & Ruprecht 1974). Diet composition was presented as the percentage of individuals consumed as prey and the percentage of consumed biomass. To calculate the biomass for each prey category, the number of prey specimens of each kind was multiplied by the mean body mass for the given prey category. The mean body mass for the prey types was as follows: rodents – 25 g, insectivores (the mole *Talpa europaea*) – 80 g, mustelids (the weasel *Mustela nivalis*) – 70 g, juvenile brown hares (*Lepus europaeus*) – 300 g, passerines – 20 g, waterfowl – 1,000 g, juvenile domestic chicken (*Gallus domesticus*) – 500 g, juvenile Hooded Crow (*Corvus corone*) – 250 g, frogs – 20 g, fish – 200 g.

Nonparametric tests were used to analyse results obtained from the comparison of: the Marsh Harrier's diet among the three nests, types of food delivered by males and females, methods of 'remains analysis' and 'pellet analysis' employed in this study. For the purpose of statistical analysis, the Marsh Harrier's diet was divided into nine food categories.

Results

In all the three nests, the Marsh Harrier's chicks were fed mainly on *Microtus* voles, which constituted about 56% of all prey items (Table 1). Common vole individuals (*Microtus arvalis*) constituted 80% of all recognised voles, while the root vole (*Microtus oeconomus*) and the short-tailed vole (*Microtus agrestis*) were hunted much less frequently. Other types of mammalian prey delivered to nestlings by adults were striped field mice (*Apodemus agrarius*), moles, weasels and juvenile brown hares. Marsh Harriers also hunted birds, small passerines and waterfowl being the most common prey. Frogs and fish constituted supplementary food items and their share in chicks' diet did not exceed 5%.

The percentage of a particular type of prey in the diet of nestlings does not reflect the actual food consumption due to the fact that it does not take into account the prey biomass. Due to their large body mass, waterfowl, although not very frequently hunted, constituted the staple food for the Marsh Harrier's chicks. At Lake Inulec, juvenile domestic chickens were sort of equivalent to waterfowl and, together with juvenile hares, constituted an important component in the diet of young Harriers. Rodents, hunted in large numbers, constituted 16.7% of the consumed prey biomass.

Despite the fact that voles were the staple food of nestlings at all the sites, the composition of the food delivered to the three nests varied significantly. Both data obtained from the analysis of collected remains and from observations of prey types delivered by parents ($\chi^2 = 49.85$, df = 16, $p < 0.001$; Fig. 1) as well as data derived from pellet analysis ($\chi^2 = 32.58$; df = 16; $p < 0.01$) confirmed that these differences were significant. Moles were preyed upon only by Harriers from Nest 2, and juvenile hares only by two pairs of Harriers at Lake Inulec (Nest 2 and 3). Passerines were hunted at all sites. Waterfowl occurred regularly in the diet of nestlings at Lake Śniardwy, but were recorded only once in Nest 2. In contrast, juvenile domestic chickens were not hunted at Lake Śniardwy, but were abundant in the diet of the Marsh Harrier's nestlings at Lake Inulec.

Table 1. Diet composition of the Marsh Harrier in the Mazurian Lakeland, NE Poland. Data pooled from the analyses of food remains and pellets from the three nests. % N – percentage of prey individuals; % B – percentage of biomass consumed.

Prey category	% N	% B
Microtus arvalis	12.32	3.01
Microtus agrestis	1.96	0.48
Microtus oeconomus	8.12	1.98
Microtus sp.	33.61	8.20
Apodemus agrarius	0.28	0.07
Apodemus sp.	0.28	0.07
Rodentia unidentified	12.04	2.94
Talpa europaea	2.80	2.19
Mustela nivalis	0.28	0.19
Lepus europaeus	1.40	4.10
Mammalia total	73.11	23.21
Podiceps cristatus	5.32	51.91
Anas platyrhynchos	0.28	2.73
Gallus domesticus	1.12	5.46
Fulica atra	1.12	10.93
Alaudidae	0.28	0.05
Luscinia sp.	0.28	0.05
Saxicola rubetra	0.28	0.05
Oriolus oriolus	0.28	0.05
Corvus corone	0.28	0.68
Emberiza citrinella	0.28	0.05
Passeriformes sp.	5.04	0.98
Aves unidentified	8.96	1.75
Aves total	23.25	74.72
Amphibia	2.24	0.44
Pisces	0.84	1.64
Total	N = 357	36,605 g

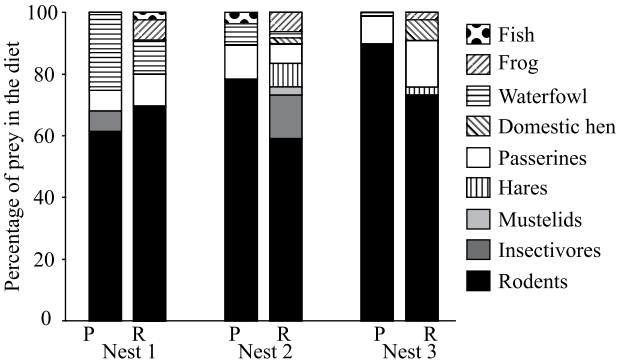


Figure 1. Diet of the Marsh Harrier's chicks depending on the investigation method: the percentage of prey in the diet obtained from pellet analysis (P) compared with the percentage of prey in the diet obtained from the analysis of food remains and visual observations of prey delivery by parents (R).

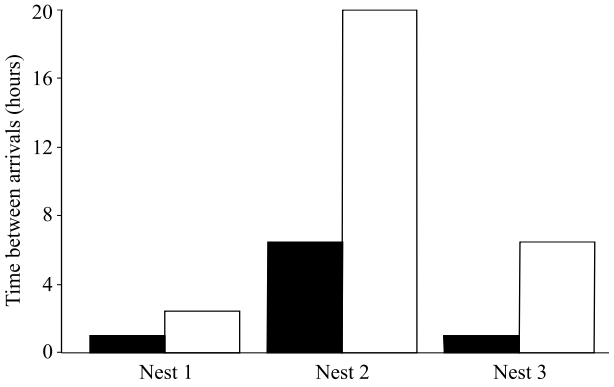


Figure 2. The average number of male (open bars) and female (filled bars) arrivals with food to each Marsh Harrier's nest per hour of observation.

At each of the three nesting sites observed, the frequency and length of parental visits varied depending on the parent's sex (Fig. 2) (Females $\chi^2 = 22.76$; df = 2; $p < 0.001$, males $\chi^2 = 14.10$; df = 2; $p < 0.001$). The frequency of female arrivals varied from one arrival per 6.7 hours to one arrival per 1.1 hours. Males visited their nests with a frequency of one arrival per 20.0 hours to one arrival per 2.3 hours.

During all observation sessions at the three nests of Marsh Harriers, a total of 281 arrivals of adult birds to the nests were recorded. Most of them (66.9%) were arrivals with food for nestlings, 21.7% were arrivals without food and in 11.4% of the cases adults arrived with plant material, which was used for fixing the nest. Females delivered food to the nests approximately three times more frequently than males ($\chi^2 = 95.5$, df = 1, $p < 0.001$).

A male, upon arrival at the nest, would usually spend just a few seconds there and after food delivery would fly away immediately. Females stayed with nestlings much longer than males ($\chi^2 = 19.4$; df = 3; $p = 0.0002$; Fig. 3).

The comparison of prey types delivered by males and by females did not reveal any significant differences ($\chi^2 = 10.3$, df = 7, $p > 0.05$). However, all domestic chickens hunted by Harriers were brought to the nests only by males while all waterfowl were delivered only by females. Females also specialised in hunting young hares.

The prey biomass given to nestlings by male and female Harriers differed in nature and amount (Fig. 4). The total biomass of rodents delivered by females was two times larger than that (the biomass composed of rodents) hunted by males. The large biomass of waterfowl hunted by females dominated the overall prey biomass. These sex-based differences in the efficacy of provisioning efforts were not so expressed in two pairs at Lake Inulec as in the Harriers' nesting site at Lake Śniardwy, where the female supplied nestlings with a large number of Coots and Grebes.

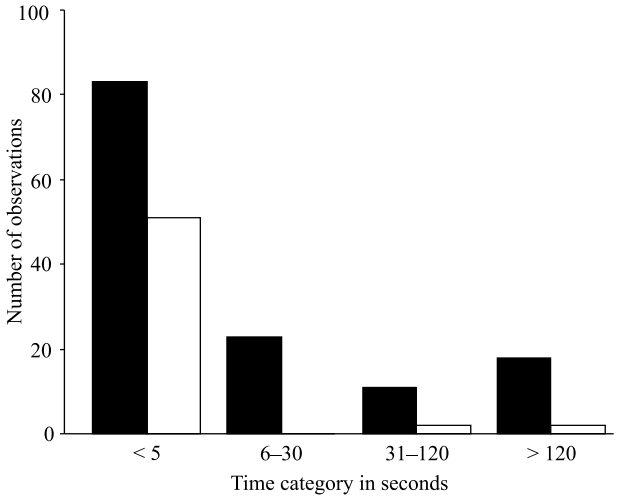


Figure 3. Time spent on nests by males (open bars) and females (filled bars).

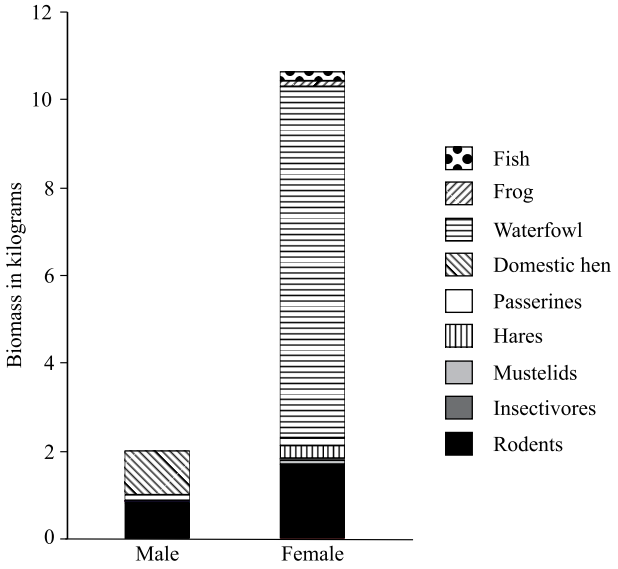


Figure 4. Biomass of prey delivered by males and females to the three Marsh Harrier's nests.

The Marsh Harrier's diet reconstructed from pellet analysis differed from that deduced from the data pooled from collected carcass analysis and observations of prey delivered by parents (Fig. 2). The differences in results obtained by the two methods were significant or nearly significant for each analysed nest (Nest 1: $\chi^2 = 14.21$; df = 5; $p = 0.014$; Nest 2: $\chi^2 = 13.58$; df = 8; $p = 0.093$ and Nest 3: $\chi^2 = 11.59$; df = 5; $p = 0.041$), as well as for the pooled data ($\chi^2 = 20.76$, df = 8, $p = 0.008$). In general, the share of rodents in the Marsh Harriers' diet reconstructed from pellet analysis was larger than in that deduced from the data pooled from carcass analysis and from observations. In contrast, remains of frogs, domestic chickens and hares were not found in pellets, although Marsh Harriers brought all these animals to the nests. In general, the diet composition based on the analysis of food remains found in the nests and from observations of prey delivery by parents is more diverse than that reconstructed from pellet analysis. Both methods showed *Microtus* voles to be the predominant prey, although the share of the four most common rodents in the diet was not the same: it varied depending on the analysis method used. The share of the common vole in the diet reconstructed from pellet analysis was significantly lower ($\chi^2 = 4.87$, df = 1; $p = 0.027$), while that of the root vole was significantly higher ($\chi^2 = 8.59$, df = 1; $p = 0.003$) when compared with the data derived from carcass analysis and observations. Percentage frequencies of *M. agrestis* and *A. agrarius* occurrence in the diet obtained by the two methods did not differ significantly.

Discussion

Results of our studies revealed the asymmetric pattern of males' and females' activity in feeding of nestlings. Prey biomass delivered by males was approximately one-fifth of that delivered by females. This is due to two factors. First, females were seen arriving with food approximately three times more frequently than males. Secondly, prey species delivered to nestlings by females were heavier than those delivered by males. However, the comparison of male and female hunting efforts and prey choices, based on observations of birds bringing food to the nest, may be biased due to the phenomenon of food transfer between the male and the female, which is common among Harriers (e.g. Wiącek 2006a). Aerial food transfer intensifies during the breeding period and, as shown by Azkona (1994), can constitute up to nearly 80% of all food transfers. In many cases, this fact makes it impossible to distinguish between prey caught by the female and prey delivered to her by the male. Thus, one must take into account that some of the prey items recorded as caught by females were in fact caught by males, whereas prey delivered by males was undoubtedly hunted on their own. The high proportion of food transferred from males to females outside the nest can explain a relatively small difference in the variety of the prey provided to chicks by both parents. Due to the difference in body size between male and female Marsh Harriers, large prey is much more accessible to females than to males. This hypothesis is partially supported by Witkowski (1989), who noted that in the Barycz River Valley, common voles constituted 63% of the prey delivered by males to incubating females. This proportion did not change significantly after the hatching of chicks, so long as prey was delivered to the nest only by the male. The number of larger prey items

delivered to young Harriers started to increase as they grew older, and the adult females joined the males in hunting efforts. Observations made by Witkowski (1989) illustrate that Marsh Harrier females and males searched for prey in different habitats and hunted different prey. In general, males specialised in hunting small terrestrial prey, mainly common voles and Skylarks, while females chiefly hunted aquatic or semi-aquatic animals such as carp, waterfowl (Coot, Grebes, Ducks) and frogs. Our study did not reveal significant differences between prey items delivered by each of the parents probably because males had previously transferred part of the hunted animals to females. On the other hand, the load size effect should also be taken into account. Small prey – below a particular energy value – is often not worth transporting to the nest over a long distance (Sonerud 1992). This rule may apply to Marsh Harrier males, who, in contrast to females, often hunt far away from the nest (Glutz *et al*. 1971; Cramp & Simmons 1980). However, during a period of particularly high food demand, one can assume that males probably deliver most of the prey they catch regardless of its size.

The number of male and female arrivals with food to the nests with four chicks was significantly higher than that to the nest with two chicks. This fact can be easily explained by higher food demands of nestlings in Nest 1 and 3. Another explanation for that could be the fact that the area inhabited by the pair with four chicks had larger food resources than that inhabited by the pair with two nestlings.

The resulting composition of the Marsh Harrier's diet, reconstructed from the analysis of food remains and from observations, differed from that deduced from pellet analysis. Similar discrepancies in results based on different methods used to study the diets of raptors were noted in the past (e.g. Redpath *et al*. 2001; Wiącek & Niedźwiedź 2005; Selås *et al*. 2007). Redpath *et al*. (2001) concluded that the share of birds that have fallen prey is under-represented and that of mammals is over-represented in the diet composition reconstructed from pellet analysis. What is more, prey remains found in the nests and the surrounding area do not fully represent the share of small species in the diet. We found amphibians to be a group of prey underestimated in pellets and this result is similar to the results reported by Selås *et al*. (2007) and Tornberg and Reif (2007) who studied food of Common Buzzards. Simmons *et al*. (1991) suggest that amphibians are always underestimated in pellets because in contrary to bones of mammals, their bones are digested and not regurgitated. Interestingly, our study illustrates that the share of very similar prey species (the common vole and the root vole) in the diet may differ depending on the analysis method used (analysis of 'pellets' *vs*. that of 'remains'). This difference was unexpected and it warrants further research.

Tornberg and Reif (2007) has concluded recently that indirect methods yield more precise results when applied for the investigation of the diet of raptors which do not specialise in a certain prey species. Meanwhile, direct methods are more reliable when studying raptors that are food specialists. However, the differentiation between specialists and generalists is not always evident and thus, it seems that to increase the reliability of results, various methods should be used for the diet analysis of raptors (Simmons *et al*. 1991; Redpath *et al*. 2001).

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Nendrinės lingės (*Circus aeruginosus*) jauniklių dieta ir jų maitinimo elgesys

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Santrauka

Trys nendrinės lingės (*Circus aeruginosus*) lizdai buvo stebimi perėjimo sezono metu prie dviejų ežerų šiaurės rytų Lenkijos Mozūrijos ežerynų rajone. Patelės atskrisdavo prie lizdų dažniau nei patinai ir praleisdavo daugiau laiko su jaunikliais. Visose lizdavietėse pagrindinę dalį laimikių, kuriuos tėvai atnešdavo savo jaunikliams, sudarė *Microtus* pelėnai (56% visų laimikių) ir kiti graužikai (12,5%). Tačiau maisto, kuriuo mito nendrinės lingės jaunikliai trijuose lizduose, sudėtis ryškiai skyrėsi. Vandens paukščiai ir naminės vištos jaunikliai sudarė 71% jauniklių suvartojamos biomasės. Tarp grobio, kurį savo jaunikliams atnešdavo nendrinės lingės patelės ir patinai, nebuvo didelio skirtumo. Tačiau kai kuriuos laimikius, kaip pavyzdžiui ausuotuosius kragus, laukius ar pilkuosius kiškius, į lizdus atnešdavo tik patelės. Patelių atnešta bendra laimikių biomasė buvo apytiksliai 5 kartus didesnė nei patinų. Mitybos sudėtis, nustatyta dvėselienos analizės bei stebėjimų pagrindu, patikimai skyrėsi nuo atrajų analizės rezultatų.

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Vegetation survey of the Coastal Grazing Marshes, and Dunes at Minsmere RSPB Nature Reserve Suffolk, 1993

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November 1996

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Summary

- The vegetation of the coastal grazing marshes (north of the New Cut), dunes and strandline at the RSPB's Minsmere Nature Reserve were surveyed in the summer of 1993.

- The 24 NVC communities identified on the grazing marshes included four associated with saltmarshes. One of these showed affinities with SM20 *Eleocharis uniglumis* saltmarsh, a west-coast community.

- In addition, eight aquatic communities were recognised in the grazing marsh ditches. They included *Spirodela polyrhiza - Hydrocharis morsus-ranae*, an increasingly scarce community, and two communities associated with coastal brackish ditches in south-eastern and southern England: A6 *Ceratophyllum submersum* and A21 *Ranunculus baudotii*. Seven pondweeds (*Potamogeton* and *Zannichellia*) were recorded in the ditches, including the Nationally Scarce *Potamogeton trichoides* and regionally rare *P. alpinus*. The Nationally Scarce *Sonchus palustris* is now widespread and apparently increasing along ditch banks on the Levels.

- A total of 14 salt-tolerant plant species occur on the grazing marshes and another two on the landward side of the dunes. Much of the saline influence appears to arise from the leaking sluice at the mouth of the New Cut, but some may result from seepage through the dunes and residual salt along the old course of the Minsmere River that was formerly tidal.

- The 16 NVC communities (plus 10 sub-communities) identified on the strandline and coastal dunes, included the nationally scarce SD1b *Rumex crispus - Glaucium flavum* shingle community *Lathyrus japonicus* sub-community which is a feature of the East Suffolk coast. As well as the *Glaucium* and *Lathyrus*, *Crambe maritimum* featured on the strandline, particularly towards the south (Sizewell Power Station) end of the reserve.

- The dune vegetation grades into acid fixed-dune grassland SD12a *Carex arenaria - Festuca ovina - Agrostis capillaris* and heathland (H8 *Calluna vulgaris - Ulex gallii*). The latter is largely associated with South-west England and Wales. *Sedum anglicum*, another characteristic west coast plant, also occurs on the Minsmere dunes.

- Three main conservation recommendations arise from the survey:
  - To fence off sections of dune and strandline to protect existing vegetation of conservation importance from further erosion by human trampling, and to allow it to recolonise. This particularly applies to the dunes north of the Sluice (Compartment 49).

  - To cut back trees, scrub and other tall vegetation shading the ditches around Compt. 41 at East Bridge. These ditches have some of the best freshwater aquatic communities on the reserve.

  - To retain the salt-tolerant plant communities and species on the Minsmere Levels by allowing brackish water into ditches near the Sluice and by grazing the fields with cattle.

  - To prevent scrub invasion at the southern end of the dunes (Compt. 59).

1 Introduction

The RSPB's Minsmere Nature Reserve on the East Suffolk coast has a variety of habitats, including 137 ha of grazing marsh and 33 ha of coastal dune and beach. Between June and September 1993, these last two habitats were surveyed. The grazing marsh is situated on the south-east side of the reserve, with 43 ha to the north of the New Cut Channel that flows out to the sea at the Sluice just behind the dune ridge (478662). This northern section of grazing marsh which incorporates eight fields and their associated ditches (Compts. 70-77 and 80) to the south of the old course of the Minsmere River was included in the survey (Figures 1-18). The ditches surrounding Compt. 41 at East Bridge were also covered (Figure 19). The grazing marshes to the south of the New Cut (94 ha) were surveyed in the same summer by Russell Cryer (1994). These latter marshes were acquired as an extension to the reserve in 1996. The beach within the reserve (Compartments 49, 58 and 59) extends from the Minsmere cliffs (TM477675) at the north end to the south end of the reserve (475645). The survey, however, extended a further 700m south of the reserve boundary to opposite the Sizewell Power Station (475638).

2 Aims

a) To identify the National Vegetation Classification (NVC) communities of the beach, dune, coastal grazing marsh and associated ditch habitats at the RSPB's Minsmere Reserve.

b) To identify botanical features of conservation importance associated with these habitats on the reserve.

c) To determine the effects of management, particularly grazing by cattle and geese, winter flooding and salinity on the vegetation of the coastal marshes and their ditches.

d) To assess the vulnerability of the beach and dune vegetation to human trampling.

3 Methods

The beach and dunes in Compt. 49, 58 and 59 at the RSPB's Minsmere reserve were surveyed between 14 and 24 June 1993. The coastal grazing marshes north of the New Cut (Compts. 70-77 and 80) were surveyed between 17 and 24 June, but the ditches in these compartments were examined between 11 August and 2 September.

The 1 km squares surveyed were TM4764, 4765 and 4766 (largely the coastal strip including the dunes), 4666 grazing marsh and a damp meadow in 4566 at East Bridge.

Before the NVC survey was undertaken, the strandline, dunes and grazing marshes were walked to gauge the communities to be sampled. With few exceptions, a minimum of five 2 x 2m quadrats were sampled for each community/sub-community. In a number of instances, an additional five quadrats were taken when a similar community was thought to occur in other fields or discrete areas. In each quadrat, percent cover frequency of each species and the average vegetation height were recorded. Constants were species recorded in each of five quadrats. Percent cover frequencies were converted to Domin values (1-10). NVC stands were mapped for each field. Details are given in the appendix.

The ditches were sampled either at the mid-point along them (short ditches) or at intervals 100m apart (longer ditches). Corners/junctions of ditches were avoided for sampling. The samples were each 20m long. Two sets of plant frequencies (percent cover) were recorded:

one for aquatics and emergents in the water, and second the bank vegetation up to 1m from the water. The depth and width of the water in the ditches were recorded along with height of the bank above water. Salinity of the water was measured using a conductivity meter.

NVC communities were identified using Rodwell (1991, 1992, 1995 and in prep.)

Results

4 Grazing Marshes

4.1 NVC communities

A total of 24 NVC communities plus an additional four sub-communities were identified on the grazing marshes within the Minsmere Reserve, north of the New Cut (Table 1). They included four associated with saltmarshes, six with swamps, two with mires, eight with mesotrophic grassland and four with aquatic situations.

4.2 Saltmarsh

The saltmarsh communities SM16, 20, 23 and 28 occurred in the fields nearest the sluice (Compt. 80) and those adjoining the old course of the Minsmere River that was formerly tidal.

SM16b *Festuca rubra* saltmarsh *Juncus gerardii* sub-community.
This occurred in almost all the fields except those furthest inland (Compts. 41 and 70). Such fields tended to be shallowly flooded in winter, and intensively grazed by cattle between May and October, and by geese for much of the year. *Juncus gerardii* occurred in all 25 2 x 2m quadrats sampled for this community and the average Domin frequency was 6.9. *Potentilla anserina* was the next most widespread species, occurring in 21 of the 25 quadrats (average Domin frequency 3.2). Other widespread species were *Agrostis stolonifera* and *Holcus lanatus* (each in 12 quadrats), *Festuca rubra* and *Lolium perenne* (each in 11). Other saltmarsh indicators were *Bolboschoenus maritimus* (Compts. 73 and 80e), *Eleocharis uniglumis* (Compts. 75 and 77). *Glaux maritima* (Compts. 71 and 75), *Salicornia ramossima* (Compt. 71) and *Aster tripolium* (ungrazed Compt. 80e).

SM16c *Festuca ribra* saltmarsh *Glaux maritima* sub-community.
A small area in the north-east corner of Compt. 71 was provisionally assigned to this community. In the one 2 x 2m quadrat *Glaux maritima* and *Salicornia ramossima* were each present at a Domin frequency of 7, with *Agrostis stolonifera* Domin 4. About 10% of the ground was bare.

SM20 *Eleocharis uniglumis* saltmarsh.
A saltmarsh community, featuring *E. uniglumis* as a constant and occurring at a Domin frequency of 8-9, occupied extensive areas in ungrazed Compt. 80 (sections a, b, d and f) nearest the Sluice. This community showed affinities with SM20 associated with west coast saltmarshes from North Wales northwards and apparently unrecorded on the east coast.

SM23 *Spergularia marina - Puccinellia distans* saltmarsh.
This community occurred as a lawn in a large, shallow depression in Compt. 72. The depression was flooded in winter and was heavily grazed by geese as well as cattle. Constants were *Spergularia marina* (average Domin 6.2) and *Glaux maritima* (7.0). *Juncus bufonius* was also frequent (IV 3.6), but *Puccinellia distans* was not recorded. Between 2 and 8% of the ground was bare. There were also two small areas of what was possibly a variant of this community in Compt. 71 grazed by cattle. Both *S. marina* and *Rumex maritimus* were frequent.

SM28 *Elytrigia repens* saltmarsh

This occurred on the slightly elevated margins and ridges in three fields (Compts. 72, 73 and 80). In Compt. 73 (five 2 x 2m quadrats sampled), constants were *Elytrigia repens* (average Domin 7.6), *Poa trivialis* (6.0) and *Potentilla anserina* (4.8). The average sward height was 74.0 cm. Compts. 73 and 80 were ungrazed; Compt. 72 was grazed by 15 cattle when surveyed.

4.3 Sand-dune

SD12a *Carex arenaria - Festuca ovina - Agrostis capillaris* fixed dune grassland. On a sandy rabbit-grazed ridge in Compt. 74.

4.4 Mesotrophic grassland

MG1 *Arrhenatherum elatius* coarse grassland

In the south-east corner of ungrazed Compt. 73 and narrow strips on higher ground in ungrazed Compt. 80 (a, f and g).

MG6a *Lolium perenne - Cynosurus cristatus* pasture

Apparently present as an intergrade with SM16b *Festuca rubra* saltmarsh, *Juncus gerardii* sub-community in areas of grazed Compt. 76. Possibly originating from a reseeding attempt. Constants *Juncus gerardii* (Domin 7), *Cynosures cristatus, Trifolium repens, T. pratensis* (all 5) and *Holcus lanatus* (4). No *Lolium* recorded.

MG7c *Lolium perenne - Alopecurus pratensis - Festuca pratensis* flood pasture

Present on a curved ridge of higher ground in ungrazed Compt. 70.

MG8 *Cynosurus cristatus - Caltha palustris* flood pasture

In wetter NE part of mown Compt. 41. Constants *Caltha palustris, Agrostis stolonifera, Holcus lanatus, Lotus pedunculatus* (all Domin 5), *Trifolium pratensis* and *Oenanthe fistulosa* (4).

MG10 *Holcus lanatus - Juncus effusus* rush pasture

a Typical sub-community: a lightly grazed sward that may be ascribed to this was present over a large area of Compt. 76.

b *Juncus inflexus* sub-community

A community in which *Holcus lanatus* (Domin 8), *Poa trivialis* and *Cirsium arvense* (both 5) were constants and *Juncus inflexus* occurred as scattered clumps was present in ungrazed Compt. 75 and grazed Compt. 76. Also in ungrazed Compt. 70 with *Carex hirta* (6) among clumps of *Juncus inflexus*. A community which *Holcus lanatus* (Domin 7) and *Cirsium arvense* (5) were constants but *Juncus inflexus* was absent and was also ascribed to MG10b in rabbit-grazed grassland in Compt. 74.

MG11b *Festuca rubra, Agrostis stolonifera, Potentilla anserina* inundation grassland

A community which had some characteristics of this occurred in lawns closely grazed by cattle and geese in Compts. 71, 72 and 73. Constants were *Juncus bufonius* (Domin 10) and *Potentilla anserina* (2). *Agrostis stolonifera* was frequent, but *Festuca rubra* was absent. The presence of *Glaux maritima* and *Spergularis marina* indicated brackish conditions. *Rumex maritimus* was associated with MG11b in Compts. 71 and 72. The areas would have been shallowly flooded in winter.

MG12a *Festuca arundinacea* coarse grassland, *Lolium perenne - Holcus lanatus* sub-community

This community was present on higher ground in Compts. 71 (on an east-west ridge), 77 and 80 (sections a, b, d, and g). In Compt.71 constants in the five 2 x 2m quadrats sampled were *Festuca pratensis* (average Domin frequency 7.2), *Poa pratensis* (3.2), *Potentilla anserina* (2.4) and *Cirsium arvense* (1.6). Also frequent was *Holcus lanatus* (IV 4.6) and *Juncus gerardii* (IV 3.4). This field was grazed by cattle (17 at the time of the survey). Geese grazed the short vegetation in the field, but not MG12a. Compts. 77 and 80 were ungrazed.

MG13 *Agrostis stolonifera - Alopecurus geniculatus* inundation grassland
This community featuring *Alopecurus geniculatus, Glyceria fluitans* and *Juncus gerardii* occurred in four wet depressions in Compt. 70, three in the north-east and one in the south-east. As an intergrade with S6, it was present in a wet runnel in the north-east corner of this field. In Compt. 80 section a, MG13 was present in a small damp hollow, perhaps the remains of a ditch.

4.5 Mires

M22 *Juncus subnodulosus - Cirsium palustre* fen meadow.
Two small patches in which *J. subnodulosus* was dominant occurred in Compt. 76 that was grazed by 24 cattle. A small area of what was possibly this community was present among SM16b saltmarsh in Compt. 75. In the SE half of mown Compt. 41.

M23a *Juncus effusus/acutiflorus - Galium palustre* rough pasture.
Small patches in wetter ground among MG10b *Holcus lanatus - Juncus effusus* rough pasture in ungrazed Compt. 75.

4.6 Swamps

S4a *Phragmites austriaca* reed-bed.
This community occurred along ditches bordering Compts. 72, 73, 75, 77 and 80, often as intergrades with other communities - S6 (with *Carex riparia*), S21 (with *Bolboschoenus*) and SM28 (with *Elytrigia repens*). It is transformed into other communities by grazing.

S6 *Carex riparia* swamp.
This formed prominent stands in the wettest parts of most fields (Compts. 70, 71, 72, 73, 76, 77 and 80 (sections a and b). Compts. 71, 73, 76 and 77 were grazed by cattle. *Carex riparia* was constant in 13 quadrats (average Domin frequency 7.7). *Cirsium arvense* was also frequent, occurring in 10 quadrats (average Domin 2.8).

S19 *Eleocharis palustris* swamp.
A small area occurred in a wet depression in Compt. 76. *Eleocharis* was associated with *Glyceria fluitans*. The Compartment was grazed by cattle.

S20 *Schoenoplectus tabernaemontani* swamp.
A stand along ditch 753 between Compts. 75 and 76. Also with *Juncus gerardii* and *Bolboschoenus maritimus* in a small wet depression on the south side of Compt. 72, near ditch 724. *S. tabernaemontani* is grazed by cattle. Probably a saltmarsh relict.

S21b *Bolboschoenus maritimus* swamp, *Agrostis stolonifera* sub-community.
Recorded along what was probably an 'extinct' channel in Compt. 71; over some extensive areas in both Compts. 72 and 73, and in Compt. 80 sections c, d and e. In Compt. 71 *Bolboschoenus* was a constant (average Domin frequency 8.0 for five 2 x 2m quadrats) along with *Agrostis stolonifera* (Domin 6.4). Close to ditch 724 on the north side of Compt. 73, *Bolboschoenus* occurred with *Phragmites* over a small area; Compts. 71 and 72 were grazed by cattle and geese; Compts. 73 and 80 were ungrazed. The presence of this community is further evidence of the estuarine origins of the Minsmere Levels.

5 Ditches

5.1 Aquatic plant communities

Eight aquatic NVC communities plus an additional sub-community were provisionally identified on the Minsmere Levels within the RSPB reserve (Table 1).

A2 *Lemna minor*

This floating community was widespread. *L. minor* and *L. trisulca* were present together in 20 ditch samples, *L. minor* alone in nine samples, and *L. trisulca* alone in six. Samples with abundant *L. trisulca* (703, 713, 722 a and b, 723 and 413) could be assigned to the *Lemna trisulca* sub-community.

A3 *Spirodela polyrhiza - Hydrocharis morsus-ranae.*

Ditch 411 in Compt. 41 beside the road at East Bridge was the only one in which both *Spirodela* and *Hydrocharis* were present. In addition there was much *Myriophyllum spicatus, Ceratophyllum demersum, Lemna minor* and formerly *Hottonia palustris.* It was partially shaded by roadside trees.

What was probably this community, with *Hydrocharis* constant (Domin 1-5) but no *Spirodela,* occurred in another six ditches in Compts. 70, 73 and 75. *Lemna minor, L. trisulca* and *Myriophyllum spicatum* was frequent. *Hydrocharis* was abundant in ditch 702b.

A5 *Ceratophyllum demersum.*

This community with abundant *Ceratophyllum* (90% cover) occurred in a shady ditch along the west side of Compt. 41 at East Bridge. Here the community was species-poor. It was also recorded in Ditch 724 (Compt. 72). A5 is usually associated with eutrophic fresh water.

A6 *Ceratophyllum submersum.*

Recorded in five or six ditches in Compts. 71, 72, 71/72 and possibly 75, 76/77 and 80. *C. submersum* was constant (Domin 4-7); *Lemna minor* and *L. trisulca* was frequent. The water where this community occurred was brackish (2.0% in most instances but 5.0% in Compt. 80). At three of the 10 sample sites where *C. submersum* was recorded, *C. demersum* was also present, but in general, the latter species was in fresh water. A6 is mostly confined to sites on or near the coast of south-east England (Rodwell 1995).

A11 *Potamogeton pectinatus - Myriophyllum spicatum.*

A community that could possibly be assigned to this occurred in shaded Ditch 413 in Compt. 41 at East Bridge. As well as having *Myriophyllum spicatum* in abundance (up to 95% cover), *Hydrocharis* and both *Potamogeton natans* and *P. alpinus* were present. The latter species is a rarity in East Anglia. *P. pectinatus* was not recorded, but was present with *Myriophyllum* in a species-rich ditch, 703 (Compt. 70).

A12 *Potamogeton pectinatus*

A species-poor community with *P. pectinatus* in two brackish ditches, 803 and 804 (Compt. 80), may tentatively be assigned to this. In the case of 803, there was an abundance of *Callitriche platycarpa* (A16?). The salinity in these ditches in September was 3-5‰.

A16 *Callitriche stagnalis.*

A community with abundant *C. obtusangula* in freshwater in the New Cut (near the bridge) may tentatively be assigned to this. *Callitriche platycarpa* occurred in three brackish peripheral ditches in Compt. 80: abundant in 801 and 803; frequent in 802. It was associated with *Ceratophyllum submersum* in 801 (A6?), with *Potamogeton pusillus* and *Ranunculus baudotii* in 802 (A21?) and with *Potamogeton pectinatus* in 803 (A11?).

A21 *Ranunculus baudotii.*

This community was present in all six east-west internal ditches (805-810) and possibly one peripheral ditch 802 in Compt. 80 near the Sluice. In these ditches the salinity readings ranged from 4.0-7.5‰ in September. In five internal ditches, *Ranunculus baudotii* (Domin 2-6) and *Hippuris vulgaris* (Domin 2-4) were constants. *Zannichellia palustris* was abundant in four of these ditches (Domin 6-9). *Potamogeton pectinatus* occurred in two internal ditches. Though ditch 802 was brackish (4.0‰), *Ranunculus baudotii* was only present in small quantity, while both *Potamogeton pusillus* and *Callitriche platycarpa* were frequent (A16?).

A21 occurs locally in brackish ditches around the coast of Britain and is frequent in Suffolk, Essex, the north Kent Marshes and around the Solvent. (Rodwell 1995).

5.2 Ditch flora

In the 53 ditch samples taken from ditches on the Minsmere Levels with the RSPB reserve (1993 boundaries), a total of 22 macrophytes, a moss and a charophyte was identified (Table 2). Most widespread were *Lemna minor* (58% of samples), *L. trisulca* (49%), *Callitriche platycarpa* (34%) and *Hydrocharis morsus-ranae* (32%). Eleven macrophytes and *Chara vulgaris* achieved cover frequencies of 20% or more in at least one ditch sample (Table). Among the six *Potamogeton* species were *P. alpinus* (two samples) and *P. trichoides* (one). *P. alpinus* has a largely northern distribution in Britain, and is a scarce plant in the lowlands of the south (Preston 1995). *P. trichoides* is considered a Nationally Scarce species (recorded from less than 100 10 km squares in Britain).

An interesting feature of the Minsmere Levels is the salinity gradient from 32‰ at the Sluice, where there is seepage of seawater into the New Cut and adjacent ditches, to freshwater 1.5 km inland. In Compt. 80, nearest the Sluice;, the salinity in the ditches ranged from 20 to 4.5‰ (bottom of water column) and 7.5 to 3‰ (surface). Further inland the salinity dropped from between 3.5 and 2.5‰ in Compt. 73 to 0.5‰ and freshwater in Compt. 70. In two pools on the landward side of the dunes to the north of the sluice salinities of 28 and 35‰ (seawater) were recorded in September after a dry period. Water may seep through the dunes from the sea into those pools. (Figures 14 and 17).

The ditch flora exhibited a response to this salinity gradient. *Ruppia maritima* was only recorded in the pool with a salinity of 35‰ (Compt. 49). *Ranunculus baudotii* (and Community A16) was restricted to Compt. 80 with late summer salinities ranging from 4.0-7.5‰. *Ceratophyllum submersum* (and Community A6) was only recorded from ditches in which the salinities lay between 2.0 and 5.0‰; it tended to be replaced by *C. demersum* at salinities of 2.0% and below. Species such as *Potamogeton pectinatus, P. pusillus, Zannichellia palustris* also occurred in the brackish ditches and appear to be tolerant of low salinities up to about 4.0‰.

Sixteen freshwater ditch samples had a mean of 4.9±2.3 aquatic macrophytes compared with a mean of 4.2±2.0 species in 21 slightly brackish ditch samples (2.0-3.0‰), and 3.3±1.0 species in 11 samples with a higher salinity (3.0-7.5%)(Table 4).

5.3 Aquatic plants of interest (Tables 2 and 3)

Callitriche spp.

In the absence of fruit, the species were tentatively identified by leaf shape. *C. platycarpa* was recorded in 18 ditch samples, including many which came from slightly brackish water. *C. obtusangula* was recorded from two freshwater sites on the New Cut and in a brackish ditch in Compt. 806.

Ceratophyllum submersum and *C. demersum*

C. submersum was recorded from 10 ditch samples, all of which were slightly brackish (2-5‰). It was locally abundant in three ditches: 712 (Compt. 71), 721 (between Compt. 71 and 72) and 722 (Compt. 72). It grew with *C. demersum* in two ditches: 721 and 801 (Compt. 80). *C. submersum* is distinguished by its brighter green and more flaccid foliage. *C. demersum* was recorded in 13 ditch samples and was locally abundant in the freshwater ditches in Compt. 41.

Chara vulgaris

All charophytes examined were identified as this species. (Moore 1986). It was widespread in the ditches in Compt. 75, 76 and 77, occurring in 13 samples and formed 20% or more cover in four.

Elodea canadensis
Only recorded in two of the 53 ditch samples: 413 (freshwater) in Compt. 41 and 702a (probably very slightly brackish 0.5‰ (Compt. 80).

Myriophyllum spicatum
Widespread in the ditches, recorded in 13 of the 53 samples and abundant in three. In freshwater and brackish water up to 2.5‰ salinity.

Potamogeton crispus
Only recorded in ditch 731 (Compt. 73) with a salinity of 2.5‰.

P. alpinus
Recorded in three sites along shady freshwater ditch 413 on the north border of Compt. 41 at East Bridge (Grid Ref TM451611). Identification confirmed by CD Preston (ITE, Monks Wood). Growing with *Myriophyllum spicatum* (frequent), *Lemna minor* (frequent) and *Hydrocharis marsus-ranae* (scarce). *P. alpinus* is an uncommon and decreasing plant in lowland, southern Britain (Preston 1995). It has not previously been recorded at Minsmere, but there are three post-1950 records near the East Suffolk coast (Simpson 1982).

P. natans
Only recorded in one freshwater ditch sample, 413c (Compt. 41) along with *P. alpinus.*

P. pectinatus
The most widespread pondweed on the RSPB's section of the Minsmere Levels, occurring in 11 of the 53 ditch samples. All were slightly brackish (0.5-6.0‰ salinity).

P. pusillus
Recorded in six ditch samples (Compts. 73, 75, 76 and 80). At least four of these were brackish (salinity 1.5-4.0‰).

P. trichoides (Nationally Scarce)
Only identified in small quantity in a newly cleaned ditch 772 (Compt. 77) where the salinity was 2.0‰. No other pondweeds occurred at the site and other aquatic macrophytes were very scarce. It is recorded as being rare in Suffolk (Simpson 1982). I observed it in a ditch that divides Compt. 30 from the west end of Compt. 31 at East Bridge (TM453664) in 1992.

Ranunculus baudotii
Restricted to seven brackish ditches in Compt. 80 near the Sluice. (Grid Ref TM475663). The salinity in these ditches ranged between 4.0 and 7.5‰. Associated with *Hippuris vulgaris* and *Zannichellia palustris.*

R. circinatus
Recorded in two ditch samples, 702 (Compt. 70) and 751 (Compt. 75). 702 was only very slightly brackish (0.5‰); 751 (not tested) was probably only marginally higher.

Ruppia maritima
Not recorded in the grazing marsh ditches, but abundant in a saline pool north of the Sluice (Compt. 49) - see under 6.2.

Spirodela polyrhiza
Only recorded in one freshwater ditch 411, Compt. 41 at East Bridge. Here it was locally abundant along with *Ceratophyllum demersum, Hydrocharis morsus-ranae, Lemna minor, Myriophyllum spicatum,* and formerly *Hottonia palustris.*

Zannichellia palustris
Apart from Compt. 80 (near the Sluice), where it was locally abundant in brackish (4.0-7.5‰ Cl) internal ditches, it was only recorded in ditch 773 when the salinity was 2.0‰

5.4 Emergent and bank plants of interest (Table 5)

Althaea officinalis
Fairly widespread on the Minsmere Levels where it grows on ditch banks and the edge of reedbeds. Recorded on the banks of the New Cut, Compts. 72 and 80 - nine (17.0%) of the 53 ditch samples. Tolerant of brackish conditions.

Aster tripolium
Only recorded where there was a distinct saline influence in Compt. 80c and in a saline runnel at the south end of Compt. 49 (477663), the north of the Sluice (see 6.2). Associated species were *Bolboschoenus maritimus* and *Juncus gerardii* (Compt. 80) and *J. gerardii* and *J. maritimus* (Compt. 49).

C. hirta
Locally frequent in a damp area in Compt. 70 (south) and at the foot of a sandy hill in Compt. 74 (west).

C. otrubae
Recorded in eight ditch-bank samples; widespread in Compts. 70, 75, 76, 77 and 80.

C. panicea
Only recorded in Compt. 41.

C. pseudocyperus
Only recorded on shaded bank of ditch 411 on the west side of Compt. 41 at East Bridge (452665).

Chenopodium rubrum
In a damp depression in the north-east corner of Compt. 70, along with halophytes: *Aster tripolium, Glaux maritima, Salicornia ramosissima* and *Spergularia marina*. Also on a ditch bank in Compt. 72. Frequent on ditch spoil, Compt. 78 between Compts. 71/72 and 76/77.

Glaux maritima
In short, grazed swards flooded in winter in Compts. 71, 72 and 73. Associated with saltmarsh communities SM16b and c, SM23 and with inundation grassland MG11b. Recorded from only one ditch bank sample - in Compt. 80, where there was saltmarsh vegetation.

Eleocharis uniglumis
Locally abundant in ungrazed Compt. 80, sections a, b, d and f (475662) in saltmarsh communities SM20 and SM28. Simpson (1982) states that the species is 'very rare and decreasing' in marshes near the Suffolk coast, but records it from Minsmere, Dunwich, Westleton (possible at the same site) and Leiston.

Juncus gerardii
Widespread in depressions flooded in winter in both grazed and ungrazed fields in Compts. 70 (NE cover), 71, 72, 73, 73 and particularly 80. Mainly associated with SM16b saltmarsh, but also occurring in S20, MG12a and MG13 communities. Recorded from eight (15.1%) of 53 ditch samples in Compts. 75 (three), 77 (one) and 80 (four).

Juncus subnodulosus
Recorded in Compt. 76 (two small patches), Compt. 75 (one patch in the centre where M22 intergraded with SM16b saltmarsh), and inland in Compt. 41 at East Bridge. It occurred in five ditch samples in Compts. 70/74, 76/77, 77 and 41.

Oenanthe fistulosa
Recorded in a single ditch sample - 703 in Compt. 70.

Rumex hydrolapathum
In nine (17.0%) of 53 ditch sample: New Cut (two), Compt. 70 (three), Compt. 74 (one) and Compt. 41 at East Bridge (three). The ditches in each instance contained freshwater.

R. maritimus
Occurred where water had lain in depressions during the winter in Compts. 71, 72 and 73. Associated with MG11b and SM23 communities. Abundant on ditch spoil thrown up onto a bank (Compt. 78) between Compts. 71/76 and 72/77. Recorded from three ditch samples.

Samolus valerandi
Widespread but in small quantity on bare ground on ditch banks. Recorded from seven (13.2%) of the 53 ditch samples in Compts. 70, 72, 73, 75, 76 (two) and 77 (two). Not associated with brackish ditches.

Schoenoplectus tabernaemontani
A few scattered stands in wet places: Compt. 72 and seven (13.2%) of the ditch samples, including 806, a brackish ditch, where locally abundant. Probably a salt-tolerant species.

Scutellaria galericulata
Only recorded on two ditch samples, both freshwater sites, in Compts. 70 and 77.

Sonchus palustris
Widespread and probably increasing on the Minsmere Levels. Recorded in six (11.3%) of the 53 ditch samples - Compts. 70, 71, 72 and at East Bridge, Compt. 41. Usually associated disturbed ground (ditch slubbings) on banks at the edge of reedbeds (S4a). A Nationally Scarce species.

Veronica anagallis-aquatica
Recorded in two ditch samples, in Compt. 74 (freshwater) and 80 (brackish 4.0‰.

6 Strandline and Sand-dunes

6.1 NVC Communities (Maps 1a-c)

Strandline and dune

SD1 *Rumex crispus - Glaucium flavum* shingle community.

SD1a Typical (*Crithmum - Crambetum*) sub-community.

Compt. 59 (four areas TM477657, 477655, 477653, 477650) and extending south of the Sizewell perimeter fence. On fine shingle and sand. A total of 225 *Crambe maritima* plants (including 126 seedlings), 57 *Glaucium* seedlings (no mature plants), 12 *Eryngium maritimum* plants, but no *Crithmum maritimum* within the RSPB reserve.

SD1b *Lathyrus japonicus* sub-community.

Compt. 59 (three areas 477657, 477655, 476650) and extending south of Sizewell perimeter fence. On sand and fine shingle, slightly higher up the beach than SD1a, on low dune

below eroded dune cliff. *Festuca rubra* and *Carex arenaria* constant; *Honckenya peploides* frequent.

SD2 *Honckenya peploides - Cakile maritima* strandline community.
Compt. 59 (two areas 477653, 477652) and extending south of Sizewell perimeter fence 477647 where it occurs with *Eryngium maritimum* (38 plants) and *Beta maritima.*
Compt. 49. Possibly this community with *Atriplex prostrata* (477664). On sand and fine shingle at the top of the beach.

SD6 *Ammophila arenaria* motile dune community.
SD6a *Elytrigia juncea* sub-community.
Compt. 49 (three sites) and Compt. 59 (at least two sites). On eroded edge of the dune, above the beach, with *Calystegia soldanella* and *Honckenya* in Compt. 49 (477664).
SD6b\c *Elytrigia juncea - Leymus arenarius* or *Leymus arenarius* sub-communities.
Compt. 59, (four sites - mostly in northern half of compartment on eroded edge of the dune, above the beach.
SD6d *Ammophila arenaria* sub-community.
Frequent along seaward edge of eroded dune at north end of Compt. 49 and intergrading with other communities/sub-communities. Not recorded in Compt. 59 where there is an abrupt demarcation (dune cliff) between the beach and dune.
SD6e *Festuca rubra* sub-community.
With *Calystegia soldanella* in at least one site in both Compts. 49 and 59. On seaward edge of eroded dunes.
SD6f *Poa pratensis* sub-community.
Scattered along the seaward edge of the dunes, especially at north end of Compt. 59. At one site with *Dryopteris dilatata.*
SD6g *Carex arenaria* sub-community.
Probably present intergrading with SD6e and f and SG10b in Compt. 59. With *Dryopteris dilatata* and *Rubus fruticosus* at one site.

SD7 *Ammophila arenaria - Festuca rubra* semi-fixed dune.
SD7a Typical sub-community.
Frequent in Compt. 49 and also present in Compt. 59.
SD7c *Ononis repens* sub-community.
Compt. 49 (one site 477664) and Compt. 59 (two sites - near Sluice 477661) and Compt. 59 (two sites - near sluice 477661, with *Galium verum, Trisetum flavescens* and *Achillea millefolium;* and 476656, with *Galium verum, Pilosella afficinarum* and *Dactylis glomerata*).

SD8 *Festuca rubra - Galium verum* fixed dune community.
SD8a Typical sub-community.
Compt. 59 (four sites at the north end, to the south of Sluice - 477660). With such plants as *Jasione montana, Vulpia bromoides* and *Trisetum flavescens.*

SD9 *Ammophila arenaria - Arrhenatherum elatius* dune grassland community.
SD9a Typical sub-community.
Compt. 59 - An area of tall grassland just south of the Sluice.

SD10 *Carex arenaria* dune community.
SD10a *Festuca rubra* sub-community.
Compt. 59 (in depression in dune ridge). *Anthoxanthum odoratum, Stellaria graminea, Ammophila, Agrostis capillaris* and *Arrhenatherum* constant along with *Festuca rubra.*
SD10b *Festuca ovina* sub-community.

Compts. 49 and 59 - a feature of the dune ridge over much of its length. With abundant *Dryopteris dilatata* on landward site (west-facing) slope of dune crest in Compt. 59. Integrading with SD6g.

SD11 *Carex arenaria - Cornicularia aculeata* community.

SD11a *Ammophila arenaria* sub-community.

Compt. 49 (at N. End 477670) - a small area close to the track on the landward site of the dune. With *Corynephorus canescens, Aira praecox, Festuca ovina,* and mosses *Campylopus introflexus* and *Polytrichum juniperinum.* Since the site is close to the junction with path along the bank across the Marsh, most vulnerable to trampling. Also two larger dry trampled areas (477669 and 477664) with such plants as *Rumex acetosella* and *Aira praecox.*

SD11b *Festuca ovina* sub-community.

Compt. 49 (477663) - dry gravelly area just north of the Sluice. With *Filago minima, Rumex acetosella, Plantago coronopus, Aira praecox, Cladonia portentosa.*

Compt. 59 (477662) - small area south of Sluice with *Aira praecox, Agrostis capillaris, Polytrichum juniperinum.*

SD12 *Carex arenaria - Festuca ovina - Agrostis capillaris* grassland community.

SD12a *Anthoxanthum oderatum* sub-community.

Compt. 49 - small area north of Sluice (477663) with *Sedum anglicum, Agrostis vinealis* and *Cladonia portentosa.*

Compt. 59 - a feature of the dry more level ground beside the main path on the landward side of the dunes from the Sluice to the Sizewell boundary fence. *Cladonia portentosa* abundant at 476656. With *Sedum anglicum* and *Silene uniflorum,* 476657.

SD12b *Holcus lanatus* sub-community.

Compt. 59 - replaces SD12a in some areas.

Neutral grassland

MG1 *Arrhenatherum elatius* coarse grassland community.

Compt. 49 - on a bank just north of the Sluice.

Compt. 59 - several large areas along the western edge where the dunes grade into the coastal grazing marsh.

Swamp

S26 *Phragmites australis* fen community.

Compt. 49 - small damp hollows near the track on landward side of dunes (476666). *Holcus lanatus* and *Vicia cracca* constant as well as *Phragmites. Agrostis stolonifera* and *Arrhenatherum elatuis* frequent. *Ophioglossum vulgatum* and *Oenanthe lachenalii* present. Integrading with MG1.

Compt. 59 - a large area on SW edge (475652).

Heath

H1 *Calluna vulgaris - Festuca ovina* heath?

Compt. 59 - very small areas at the extreme SW of the compartment where scrub is invading the dunes. Two clumps of *Calluna* associated with *Erica cinerea* (476650) and a third small patch (476649).

H8 *Calluna vulgaris - Ulex gallii* heath community.

H8a Typical sub-community.

Compt. 49 - some very small and scattered patches featuring *Erica cinerea* and *Ulex gallii,* but *Calluna vulgaris* absent.

Saltmarsh

SM2 *Ruppia maritima* saltmarsh community.
Compt. 49 - shallow pool north of the Sluice (477663).

SM16 *Festuca rubra* saltmarsh community.
SM16b *Juncus gerardii* sub-community.
Compt. 49 - on both (477663). With *Agrostis stolonifera, Glaux maritima, Spergularia marina* and *Atriplex prostrata*, as well as *Juncus gerardii*.

SM18 *Juncus maritimus* saltmarsh community.
Compt. 49 - Margin of saline pool and saline runnel north of the Sluice (477663). With *Aster tripolium, Juncus gerardii, Elytrigia atherica* and *Festuca rubra*, as well as *Juncus maritimus*.

6.2 Some of the more interesting plants of the strandline and dune

(*Nationally Scarce species)

Aster tripolium - saline runnel at South end of Compt. 49, near the sluice (477663).

Cakile maritima - strandline SD2, Compt. 59 (4765).

Calluna vulgaris - two clumps H1?, Compt. 59 (476650).

Calystegia soldanella - eroded edge of dunes above beach SD6a & e.
Compt. 49, (477664); Compt. 59, (476651).

Campanula rotundifolia - SD10b and SD12, Compt. 59, three sites in small quantity (4765).

**Corynephorus canescens* - dry slope on landward side of dunes
SD11 Compt. 49)477670). Very local and vulnerable to trampling. Also at N. (Dunwich) end of beach outside reserve (477675).
Introduced from Norfolk stock after the 1953 floods (Simpson 1982).

Crambe maritima - strandline SD1a. Compt. 59 (476657(A)), (476656(B)), (476653(C)) and (476650(D)). Also outside reserve, S. of Sizewell boundary fence (E).

Size of plants		Small	Medium	Large	Total
RSPB	A	44	-	18	62
	B	8	13	5	26
	C	70	19	26	115
	D	4	5	13	22
Total		126 (56%)	37 (16%)	62 (28%)	225
Sizewell E		29	10	10	49

Crithmum maritimum - strandline SH1a. Compt. 59 (476648).
(South of Sizewell boundary force, outside reserve), one plant

Cynoglossum officinale - rough grassland Compt. 59 (477660).

Deschampsia flexuosa - SD10b, Compt. 59 (477653). One small patch.

Dryopteris dilatata - locally frequent on west side of dune crest, SD6f and g, Compt. 59 (4765).

Elytrigia atherica - margin of saline pool, SM18. Compt. 49 (477663); rough grassland SD11b (477660), and landward dune SD12 (477653), Compt. 59.

Elytrigia juncea - eroded edge of dunes SD6a, b and c. Compts. 49 and 59 in a number of sites.

Erica cinerea - dry dune, H8a, three small patches with *Ulex gallii*; another two without *U. gallii*. Compt. 49 (4766); also two small clumps associated with *Calluna*, H1? at south end Compt. 59 (476650).

Eryngium maritimum - 12 plants on beach strandline, SD1a, at south end of Compt. 59 (477648). Another 38 plants, SD2, south of the Sizewell perimeter fence, outside the reserve (477647).

Filago minima - on dry gravelly E facing bank, SD11b, just north of the Sluice, Compt. 49 (477662).

Glaucium flavum - 57 seedlings on strandline, SD1a, Compt. 59 (477653). Two flowering plants and one seedling south of the Sizewell perimeter fence, outside the reserve. Also a few small plants beside track at north end of beach (Dunwich end), outside reserve (477675).

Glaux maritima - with other saltmarsh plants, saline seepage on path, SM16b, Compt. 49 (477663).

Humulus lupulus - scrub on the site of a cottage behind the dunes, near the Sluice, Compt. 59 (477660).

Jasione montana - several sites on dry dunes, SD8, SD11a and SD12, where vegetation short, Compt. 49 (one site) and Compt. 59 (4764, 4765, 4766).

Juncus gerardii - saline seepages along path and nearby runnel, with other saltmarsh plants, SM16b and SM18, Compt. 49 (477663).

Juncus maritimus - margin of saline pool and in a nearby runnel, SM18, Compt. 49 (477663).

**Lathyrus japonicus* - on sand and fine shingle at the top of the beech, SD1b, in three areas in Compt. 59 (477657, 477655 and 477650) and extending south of the Sizewell perimeter fence. Northern sites vulnerable to dune erosion. In 1970s extended into Compt. 49. The East Suffolk coast population is large and of considerable conservation importance in European terms.

Leymus arenarius - near eroded edge of the dunes, SD6b/c, at four sites in Compt. 59 (4765).

Lupinus arboreus - in some quantity, SD9, just south of the Sluice Compt. 59, mostly in 4766, but extending into 4765.

Oenanthe lachenalii - in damp hollow with *Phragmites*, S26, near path behind the dunes, Compt. 49 (477665).

Ophioglossum vulgatum - in damp hollow with *Phragmites*, S26, near path behind the dunes, Compt. 49 (477666). Has persisted at the site since at least the mid-1970s.

Orobanche minor - on dry gravelly bank, SD11b, just north of the Sluice, Compt. 49 (477661).

Plantago coronopus - frequent in trampled areas such as paths in Compts. 49 and 59.

Polypodium vulgare on west side of dune ridge, SD10b, Compt. 59 (477657).

Ruppia maritima - abundant in saline pool, SM2, north of the Sluice, Compt. 49 (477663).

Sedum anglicum - dry bare areas, SD11 and SD12, Compt. 49 (477662) and Compt. 59 (477658, 477648).

Silene uniflorum - in small quantity, SD12, Compt. 59 (477657). Also one plant just south of the Sizewell perimeter fence (477648) outside the reserve.

Smyrnium olusatrum - rough grassland just south of the Sluice, Compt. 59 (477660).

Spergularia marina - saline seepage on path, SM16b, Compt. 49 (477663).

Stellaria graminea - locally plentiful in hollows in dunes, SD10b, Compt. 59. (477653).

Trifolium arvense - dry, rather bare ground, SD11, close to the Sizewell perimeter fence at south end of Compt. 59 (477648).

Ulex gallii - four small patches on landward margin of dunes, H8a, northern half of Compt. 49 (4766).

Vulpia bromoides - on dry dune, SD8, Compt. 59 (477660).

6.3 Trees and shrubs on the Minsmere dunes

(*Probably bird 'sown'; +Planted).

Betula pendula - invading the West side of Compt. 59 from woodland to SW.

*Crataegus monogyna** - one bush, south half of Compt. 59.

Fraxinus excelsior - One small tree in thicket at NW side of Compt. 59 (site of former cottage).

*Lonicera periclymenum** - scrub north half of Compt. 49, and thicket NW side of Compt. 59.

Malus domestica+ - thicket as above, Compt. 59.

Pinus sylvestris - one bush on dunes, north half of Compt. 59, and invading SW side of that Compt.

Prunus domestica+ - thicket as above, Compt. 59.

*Quercus robur** - two small trees at south end of Compt. 59.

*Rosa canina** - one bush south half of Compt. 49; two sites south end of Compt. 59.

*Rosa tomentosa** - one bush on dunes, south half Compt. 59.

Rosa setigera+ - a bush in thicket, the site of a former cottage, NW side Compt. 59 (477660).

Rubus fruticosus agg - thickets on west side of dunes, north half Compt. 49.

Salix caprea - thicket as above, Compt. 59.

*Sambucus nigra** - scrub, north half of Compt. 49 and North end of Compt. 59.

Sarothamnus scoparius - bank north of Sluice, Compt. 49.

*Sorbus aucuparia** - one bush south half of Compt. 59.

Tamarix gallica[+] - thicket to east of path at North end of Compt. 59.

Ulex europaea - many fragmented thickets on west side of dunes in Compt. 49 and north half of Compt. 59.

7 Botanical features of conservation importance

The RSPB's Minsmere reserve is recognised as having a wide variety of habitats and hence a rich biodiversity. The survey revealed 16 NVC communities plus 10 sub-communities on the beach strandline and coastal dunes (33 ha), and 29 communities plus five sub-communities on the grazing marshes and associated ditches north of the New Cut (43 ha).

7.1 Coastal grazing marshes

An interesting feature of the vegetation of these marshes is the saline influence near the coast. There seems to be a declining salinity gradient from 30.5‰ at the mouth of the New Cut, where seawater seeps into the channel through the Sluice, to virtually freshwater in the Cut and ditches 1.5 km inland. The water in the ditches of Compt. 80, close to the Sluice had a surface salinity of between 3.0 and 7.5‰ (up to 20‰ on the bottom). It is suspected that there is some saltwater seepage under the dunes and residue salt in the soils close to the former course of the Minsmere River.

The brackish conditions in Compts. 80 and others such as 75 and 77 have enabled saltmarsh NVC communities (SM16, 20, 21b and 23) to persist. A community resembling SM20 *Eleocharis uniglumis* saltmarsh in Compt. 80 near the Sluice was unexpected since it is largely associated with saltmarshes on the north western coast of Britain. The brackish ditches in Compt. 80 have the aquatic A21 *Ranunculus baudotii* community which has a local distribution in Britain (coasts of south-eastern and southern England). Another aquatic community, A6 *Ceratophyllum submersum*, that was recorded in five or six ditches (Compts. 71, 72, 80 and possibly others), is generally associated with brackish ditches and largely occurs near the coast of south-east England.

The following 14 'halophytic' plants (tolerant of saline conditions) were recorded on the coastal grazing marshes of the Minsmere Levels north of the New Cut:

*Althaea officinalis**
Aster tripolium
*Bulboschoenus maritimus**
*Ceratophyllum submersum**
*Eleocharis uniglumis**
*Elytrigia repens**
Glaux maritima
Juncus gerardii
*Potamogeton pectinatus**
Ranunculus baudotii
Salicarnia ramosissima
Spergularia marina
*Schoenoplectus tabernaemontani**
*Zannichellia palustris**

Juncus maritimus and *Ruppia maritima* occur at a saline pool at the landwardside of the dunes, just north of the Sluice (Compt. 49).

(*Species which also grow naturally in the absence of saline conditions.)

Of the freshwater ditches, the most botanically interesting ones were those in Compt. 41 at East Bridge. In one of these ditches, the increasingly local aquatic community A3 *Spirodela polyrhiza - Hydrocharis morsus-ranae* occurred. *Hydrocharis* was a constant in the absence of *Spirodela* in six ditch samples elsewhere on the Levels north of the New Cut.

Six species of *Potamogeton* and *Zannichellia palustris* were recorded in ditches on the Levels. *P. trichoides*, a Nationally Scarce species, was recorded in one ditch in Compt. 77. In 1992, it was recorded on the reserve near East Bridge (Compt. 30/31). A new species for the reserve was *P. alpinus* in a ditch on the north border of Compt. 41, at East Bridge. It is a scarce plant in lowland, southern Britain, but not Nationally Scarce.

Sonchus palustris is Nationally Scarce; it is now widespread and apparently increasing on the Minsmere Levels, particularly associated with distributed ground along ungrazed ditch banks. Being a tall plant (over 2m) it can complete with *Phragmites*.

7.2 Strandline and dunes

Within a transect of less than 200m, across the dunes south of the Sluice (Compt. 59) it is possible to pass through up to eight sand dune communities. Towards the south end of Compt. 59 the strandline vegetation has remained reasonably intact, including some small patches of the Nationally Scarce SD1b *Rumex crispus - Glacium flavum* shingle community *Lathyrus japonicus* sub-community. In this area *Crambe maritima, Eryngium maritimum, Glaucium flavum* as well as *L. japnicus* occur. The East Suffolk coast population of *L. japonicus* is large and of conservation importance in West Europe.

The dune vegetation grades into what is virtually acid grassland (SD12a *Carex arenaria - Festuca ovina - Agrostis capillaris*) and heath (H8 *Calluna vulgaris - Ulex gallii*). *Cladonia portentosa* is locally abundant, and both *Ulex gallii* and *Sedum anglicum*, more typical of western Britain, occur.

Nationally Scarce plants that occur on Minsmere's dunes and strandline are: *Lathyrus japonicus* and *Corynephorus canescens*, though the latter was apparently introduced in the 1950s.

8 Some management recommendations arising from the survey

8.1 Strandline and dune erosion

The strandline flora which includes the Nationally Scarce SD1b community with *Lathyrus japonicus* is under considerable threat from erosion at Minsmere. It has all but disappeared from Compt. 49, to the north of the Sluice, and in Compt. 59 is now largely confined to near the southern boundary of the reserve. The *Lathyrus* was formerly more widespread (in the 1970s it was present in Compt. 49) but now only occurs in three restricted areas in Compt. 59. Much of the erosion which is cutting away the seaward face of the dune seems to be due to wave action, but in Compt. 49, to the north of the Sluice, human trampling pressure has seriously damaged the dune vegetation. Particularly at risk is the Nationally Scarce grass *Corynephorus canescens* that grows on the landward slope of the dune, almost opposite the beach exit of the footpath that cuts across the marsh from the car park and centre. Localised light to moderate trampling pressure can, however, benefit certain small plants that are out-competed by taller vegetation.

Recommendation: that sections of dunes in Compt. 49 be fenced/roped off to protect existing vegetation of conservation importance (eg *Corynephorus canescens*) from erosion by human trampling, and to allow re-establishment of strandline vegetation, particularly such species as *Crambe maritima, Glaucium flavum* and *Lathyrus japonicus*. Key areas for action are the north end (access from Dunwich cliffs) and opposite the exit of footpath across the marsh to the beach. It may be necessary to reintroduce the above plants if they do not reappear after several years of beach protection.

8.2 Ditch management

Most of the ditches on the Levels to the north of the New Cut had been recently cleaned out and were in good condition for aquatic plants. However, those around Compt. 41 at East Bridge , which were among the highest in botanical quality of the ditches surveyed, were shaded by willows and alders, and were in danger of losing their interest. This applied particularly to ditch 413 in which *Potamogeton alpinus* occurred.

Recommendation: that trees, scrub and other tall bank vegetation (eg *Carex riparia*) are cut back to allow more light to the ditches around Compt. 41. It may also be necessary to clean out sections of these ditches.

8.3 Conservation of saltmarsh and brackish ditches

An interesting feature of the Minsmere Levels is the brackish state of the ditches and fields in the vicinity of the sluice and the old course of the Minsmere River. The Sluice leaks allowing seawater to seep into the New Cut. The Minsmere River was probably tidal near its mouth in former days. There may be some seepage through the dunes. There is something of a salinity gradient as one goes inland and away from the Sluice and Minsmere River. Fifteen plant species recorded on the grazing marshes and in the ditches on the Levels were considered to be halophytes (salt tolerant). Some of the wet fields (eg 71 and 72) which are grazed by cattle are also much used by feeding geese. Conversion to a reedbed would destroy much of the botanical interest.

Recommendation: that Compts. 70-73, 75-77 and particularly 80 are retained as coastal grazing marsh which is allowed to flood in winter. This should maintain the brackish saltmarsh conditions and associated flora.

8.4 Scrub invasion of the dunes

There was evidence of birch *Betula pendula* and Scots pine *Pinus sylvestris* and even oak *Quercus robur* invading the landward side of the dunes at the south end of Compt. 59. In time this scrub could succeed the acid grassland (SD12a) and heath (H8).

Recommendation: that scrub invasion of the dunes is kept under observation and in time it may be necessary to clear it.

9 Acknowledgements

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Repeat Vegetation Survey of the Beach and Dunes at Minsmere RSPB Nature reserve, Suffolk, 2004

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Repeat Vegetation Survey of the Beach and Dunes at Minsmere RSPB Nature Reserve, Suffolk, 2004. CJ. Cadbury

Summary

The beach and dunes at Minsmere RSPB nature reserve, Suffolk have been increasingly suffering from erosion and are vulnerable to sea level rise. As a result, steps to strengthen the sea defences are being considered.

To asses the botanical conservation importance of the dune flora and the potential implications of damage to it from erosion and engineering works, a survey of the vegetation of the beach and dunes was undertaken in September 2004, repeating a similar survey in 1993.

This survey revealed encouraging recolonization by both Sea Kale *Crambe maritima* and Yellow-Horned Poppy *Glaucium flavum* of the beach north of the Sluice (Compartment 49); recolonization of the semi-fixed dune north of the Sluice by Sea Pea *Lathyrus japonicus* (NSc); strong populations of both Sea Kale and Sea Pea on an extensive fine shingle platform at the south end of the reserve (Compt 59), and two flourishing colonies of Grey Hair-grass *Corynephorus canescens* (Near Threatened) on the landward side of the dunes in Compt 49. Those sites in Compt 49 are vulnerable to both erosion and engineering works. In Compt 59, where the dunes are both wider and higher, there is erosion of the dune cliff, but strengthening of the sea defences appears less urgent.

The predominant vegetation of the fixed dune is Sand Sedge *Carex arenaria* and Marram *Ammophila arenaria* dominated (SD10) on the higher seaward ridge and Sheep's Fescue *Festuca ovina* and Sand Sedge (SD12a) on the inner landward areas. There are small fragments of a heathland type vegetation (H1 and H8), but insufficient to justify open public access. Gorse *Ulex europaeus* cover on the dune, especially to the north of the Sluice, has apparently increased since 1993. To the south of the Sluice, Sand Sedge cover has increased, particularly to the exclusion of less competitive plants. In an attempt to reduce the density and height of Sand Sedge and to control scrub invasion (mostly Pedunculate Oak *Quercus robur* and Downy Birch *Betula pubescens*) seasonal sheep grazing has been introduced at the south end of the dunes. The effect shows promise.

1 Introduction

The beach and dunes that extend along the shore of Minsmere Haven, East Suffolk, are one of the important habitats represented on the Minsmere RSPB nature reserve. They also provide an essential defence for the freshwater Minsmere Levels against sea incursion. Already there are abundant signs of erosion of the beach and dune cliff. In the winter of 2003/04 the sea over-topped the dunes at the Dunwich (National Trust) end. For the Minsmere marshes to survive in their present form it is clear that the sea defences will probably have to be strengthened, even if the measures may only be effective in the short term.

A survey carried out by the author in July 1993 had identified the botanical riches of these narrow dunes which extended from the southern end of the Dunwich cliffs to Sizewell. This survey aimed at gauging the subsequent changes in shore profile and how engineering works to strengthen the sea defences might effect the vegetation of the beach and dunes. It enables an assessment to be made of the botanical conservation importance of the Minsmere dunes, and suggests how damage to the flora could be mitigated.

2 Method

The present survey was undertaken over two days, 8 and 9 September 2004 in fine weather. The following were recorded:

a) Beach profile and substrate.
b) Height and condition (extent of erosion) of the dune cliff.
c) NVC communities of the beach and dunes back to the eastern margin of the grazing marsh.
d) Distribution of selected, less widespread plants.
e) Population size (counts of individual plants) of key strandline plants:
   Yellow-horned Poppy *Glaucium flavum*
   Sea Kale *Crambe maritima*
   Sea Holly *Eryngium maritimum*
   Sea Pea *Lathyrus japonicus (NSC)*
   Rock Samphire *Crithmum maritimum*
   Also Grey Hair-grass *Corynephorus canescens* (Near Threatened)
f) A small amount of data were collected on the effects of sheep grazing by comparing the height of vegetation, both within and outside fence lines of enclosures.

Global Positioning System was used extensively to record eight figure grid references. Data were recorded on 1:10,000 maps. Comparisons could be made with similar information collected 11 years earlier.

3 Results

Six habitats were represented on the Minsmere dunes:

a) Beach strand line and eroded dune cliff.
b) Mobile and semi-fixed dune
c) Fixed dune
d) Heathland
e) Saltmarsh

f) Scrub and woodland
The last three were only minor components.

3.1 NVC Communities (see Rodwell 2000) – Maps 1 and 3

3.1.1 Beach Strandline

In view of the changes in beach profile and erosion the strandline vegetation is likely to be particularly dynamic.

SD1 *Rumex crispus – Glaucium flavum* shingle
Yellow-horned Poppy *Glaucium flavum* was locally frequent with Sea Kale *Crambe maritima* and Sea Purslane *Honckenya peploides* on the fine shingle of the upper beach in the southern half of Compt 49. Elsewhere, however, on the shingle platform at the south end of Compt 59, where *Crambe, Honckenya* and Curled Dock *Rumex crispus* were a feature, *Glaucium* was sparse. Since Sea Pea *Lathyrus japonicus* was frequent, the community there may be assigned to the *Lathyrus japonicus* sub-community (SD1b).

(SD2 *Honckenya peploides – Cakile maritima* strandline)
With Sea Rocket *Cakile maritima* absent this community was probably not represented on the Minsmere shoreline.

3.1.2 Mobile, semi-fixed and fixed dune

SD6 *Ammophila arenaria* mobile dune community.
This Marram Grass community occurred on the seaward margin of the dunes and on the dune cliff, often where there was erosion. It was widespread, but in most places was present as a narrow band. SD6 was a feature of the upper beach north of the Sluice in Compt 49 and just south of the Sluice in Compt 59. Associates included Sea Bindweed *Calystegia soldanella* , Sand Couch *Elytrigia juncea,* Rush-leaved Fescue *Festuca arenaria* and Lyme Grass *Leymus arenarius*. None of them was present in quantity justifying SD4 *Elytrigia (Elymus) juncea* foredune or SD5 *Leymus arenarius* mobile dune communities. Tree lupin *Lupinus arboreus* was naturalized along the dune cliff just S of the Sluice in Compt 54.

SD7 *Ammophila arenaria – Festuca rubra* semi-fixed dune community.
This community formed the vegetation along the seaward fringe of the dune ridge above and on the sand cliff. Red Fescue *Festuca rubra* was frequent. In several places the Common Rest Harrow *Ononis repens* sub-community SD7c was associated with *Lathyrus japonicus* as at 4771 6555 in Compt 59.

SD10 *Carex arenaria* dune community.
This was one of the most frequent communities in Compt 59 where it extended over much of the length of the dunes on the landward (west) side of the main dune ridge. It featured a dense sward of Sand Sedge *Carex arenaria*. Associates were *Ammophila arenaria,* Spreading Meadow-grass *Poa humilis* and somewhat surprisingly Sea Couch *Elytrigia atherica*. Broad Buckler Fern *Dryopteris dilatata* was frequent in the landward side of the dune crest where there may be a more humid microclimate. *Carex arenaria* appears to have increased since 1993 and may be reducing the floral diversity of the dunes.

SD11 *Carex arenaria – Cornicularia aculeata* dune community.
This community was associated with the desiccated landward slope of the fixed dunes and was apparently restricted to a few sites in Compt 49 and the National Trust land at the north end of the dunes. Associates included Grey Hair-grass *Corynephorus canescens* (Near Threatened) in two areas (4779 6744 and 47778 6700), Sheep's Sorrel *Rumex acetosella*, Buck's-horn Pantain *Plantago coronopus*, Biting Stonecrop *Sedum acre*, English Stonecrop *S anglicum*, Sheep's-bit *Jasione montana*, Small Cudweed *Filago minima*, Cat's-ear *Hypochaeris radicata*, Early Hair-grass *Aira praecox*, Sheep's Fescue *Festuca ovina*, the mosses *Campylopus introflexus* and *Polytrichum juniperum*, and *Cladonia* lichens. The sites with *Corynephorus* were asigned to the SD11c *Ammophila* sub-community.

SD12 *Carex arenaria – Festuca ovina – Agrostis capillaris* dune grassland.
A community of the lower, drier areas on the landward side of the dunes, extending over much of the length of Compt 59. It forms a zone inland of AD10 and is considerably more open than that community *Festuca ovina* was dominant and both *Carex arenaria* and Common Bent *Agrostis capillaris* were frequent. There were areas (such as 4766 6563 and 4765 6465) where *Cladonia portentosa* was a feature. Mouse-ear Hawkweed *Pilosella officinarum*, Field Woodrush *Luzula campestris* and the moss *Dicranum scoparium* were associates. At one site (4766 6565) there was a small (relict?) patch Sea Campion *Silene uniflora* . In view of the scarcity of Yorkshire Fog, *Holcus lanatus*, the sub-community was probably SD12a.

3.1.3 Neutral grassland

MG1 *Arrhenatherum elatius* coarse grassland community
This tall grass community occurred in several places around the Sluice (Compt 59) on the landward side of the dunes. It features False oat *Arrhenatherum elatius*, Sea Couch *Elytrigia atherica* and Cock's foot *Dactylis glomerata*.

3.1.4 Heathland

Only fragments of heath occurred on the dunes among SD10 and SD12a.

H1 *Calluna vulgaris – Festuca ovina* heath
A fairly large patch of Heather *Calluna vulgaris* at the north end of the oak/birch woodland (4571 6489) probably represented this community. It was associated with an adjacent stand of Wavy Hair-grass *Deschampsia flexuosa*. Close by was SD12a dune grassland. A small patch of what could be described as H1 occurred among SD10 at 4758 6502.

H8a *Calluna vulgaris – Ulex gallii* heath
Very small and scattered fragments of what was probably this community featuring Bell Heather *Erica cinerea* and Western Gorse *Ulex gallii* occurred in Compt 49 (478 666 and 478 668). *Calluna* was in most instances absent. The community had a similar distribution in 1993.

The dunes have been proposed as an open access area on account of the presence of heathland with Heather. As shown above this is unjustified and the status should be reviewed.

3.1.5 Swamp

S26 *Phragmites australis* fen community

This occurred as a few small stands on the landward side of the dunes in Compt 49 (4776 6647) and a more extensive one on the western margin of Compt 59 abutting the grazing marsh (4772 6590). At 4776 6647 an associate was Parsley Water-dropwort *Oenanthe lachenalii*, but Adder's-tongue *Ophioglossum vulgatum*, observed there in 1993, was not refound. (This site was drier and had become invaded by *Rubus fruticosus*).

3.1.6 Saltmarsh

Saline seepage just north of the Sluice, Compt 49 has resulted in a local development of saltmarsh vegetation around two brackish pools and along the adjacent path.

SM2 *Ruppia maritima* saltmarsh

A brackish pool with abundant *Ruppia* just north of the Sluice, Compt 42. 47756631

SM16b *Festuca rubra* salt-marsh *Juncus gerardii* sub-community

This community with Saltmarsh Rush *Juncus gerardii*, Sea milkwort *Glaux maritima* and Creeping Bent *Agrostis stolonifera* predominant was a feature of runnels in the trampled path to the north of the Sluice (4775 6647 – 4775 6639). Red Fescue *Festuca rubra* was apparently absent. Silver Weed *Potentilla anserina* occurred in one section of the path.

SM18 *Juncus maritimus* saltmarsh community

This was restricted to the margins of two brackish pools and a hollow (former pit?) just north of the Sluice, Compt 49 (4775 6631 and 4775 6635). Sea Rush *Juncas maritimus* was predominant, with Sea Aster *Aster tripoleum* (ray-flowered form), Sea Couch *Elytrigia atherica* and *Agrostis stolonifera* as associates.

SM24 *Elytrigia atherica* salt-marsh community

This too, with Sea Couch *E atherica* predominant, formed stands along side SM18 around the two brackish pools just north of the Sluice. Compt 49. There were also isolated stands elsewhere, as at 4772 6604, Compt 59, where there were no obvious brackish incursions

3.1.7 Woodland and scrub

W16a *Quercus* spp – *Betula* spp – *Deschampsia flexuosa* woodland.

At the southern end of Compt 59, Pedunculate Oak *Quercus robur* and Downy Birch *Betula pubescens* had invaded SM12a dune grassland and become dense on the western margin. Wavy Hair-grass *Deschampsia flexuosa* was however decidedly scarce (see above under H1). Grazing by four or five rams in an enclosure to the north of the dense woodland had effectively killed much of the oak and young birch, but the larger birch had survived. The grazed area has reverted to SM10 with much *Carex arenaria*.

A few small stands of W16a who occur on the inner dune ridge further north.

W23 *Ulex europaeus – Rubus fruticosus* scrub

This community was widespread on the landward side of the dunes in both Compt 49 and 59. Common Gorse *Ulex europaeus* and Bramble *Rubus fruticosus* agg. were predominant, but associates were Honeysuckle *Lonicera periclymenm*, Broom *Cytisus scoparius* (scarce) and Elder *Sambucus nigra*. At 4768 6522, where the high dune cliff had been eroded, blocks of

dune with both *Lonicera* and *Rubus fruticosus* had fallen onto the beach below. The stands of *Ulex europaeus* appear to have become more extensive, particularly in Compt 49.

4 Distribution and populations of selected plants

4.1 Strand line and mobile/semi-fixed dune (see Maps 2 and 4)

Armeria maritima Thrift
A small patch on the landward slope of the eroded dune at the N (Dunwich) end TM 4778 6746. National Trust. Not recorded 1993.

Beta maritima Sea Beet
A single plant on eroded shingle and sand at N end of dunes 4779 6734. National Trust. Seven plants on a shingle platform at extreme S end, 4764 6452, Compt 59. SD1. Not recorded 1993.

(*Cakile maritima* Sea Rocket)
Recorded at 4765, Compt 59, 1993. Not observed in 2004.

Calystegia soldanella Sea Bindweed
Scattered along seaward edge of dunes, often on eroded cliff above beach. SD1, SD6, SD7.
Compt 49 4781 6716, 4781 6660, 4780 6640, 4780 6636, 4780 6633, 4780 6628.
Compt 59 4779 6614, 4774 6570, 4766 6486, 4764 6452 (shingle platform)
Probably little change from 1993, though habitat subject to erosion.

Crambe maritima Sea Kale (not scarce)
Has spread considerably since 1993 with many plants on the shingle and sand beach in Compt 49. Frequent on shingle platform at S end of Compt 59. Many young plants. SD1

Compt 49 (134 plants)		Compt 59 (267 plants)	
4779 6756	1 (small)	4779 6614	2
4782 6700	4	4777 6592	1
4781 6682	25	4777 6590	1
4782 6678	20	4776 6586	1
4782 6674	9	4772 6573	2
4781 6660	32	4772 6553	1 low cliff
4782 6659	12	4771 6550	2 shelf
4782 6654	6	4771 6545	5 step
4781 6646	7	4770 6543	2 step
4781 6638	3	4770 6541	1
4781 6633	4	4767 6495	14
4780 6628	2	4766 6488	1
4780 6626	3	4766 6476	27 sand with little shingle at base of cliff
4779 6615	5	4766 6479	20 inc one on bank
		4766 6477	1 top of cliff

4766 6475 – 7464 6452	186 fine shingle platform 292m long, 4m wide a N end, 11m at S end.

This population extends further S along the Sizewell beach. Has benefited from the change to more gentle beach profile in Compt 49 to N of the Sluice. Absent from the beach where there is a high sand cliff (2 – 5m) 4776 6536 – 4767 6501 in Compt 59. The many young non-flowering plants indicates an expanding population. The large fruits are buoyant in water and appear suitable for dispersed by sea. The dried dead plants may also be blown along the beach in winter.

Crithmum maritimum Rock samphire
Still a scarce plant in the reserve. Only three plants at 4778 6606, Compt 59. SD1. Not observed on the shingle platform at the S end of the reserve.

Elytrigia juncea Sea couch
Apparently thinly scattered on sandy foredune/strand line, but probably overlooked at this time of year. SD6 (SD1)

Compt 49	4780 6666
Compt 59	4779 6614 Eroded cliff near Sluice
	4764 6452 Fine shingle platform at S end.

Eryngium maritimum Sea Holly
Has spread and increased within the reserve benefiting from the deposition of fine shingle and sand along the beach. SD1. Only a single plant in Compt 49 N of the Sluice (4781 6633). 59 plants in Compt 59.
Compt 59:

4778 6606	1
4778 6604	5 non-flowering
4779 6600	2
4778 6597	1 nf
4777 6594	8
4777 6590	2
4776 6588	1
4776 6583	3 (1 nf)
4776 6475 – 4764 6457	26 (some nf) fine shingle platform

The latter area had 12 plants in 1993.
The population extends S towards Sizewell, but was not surveyed.

Festuca arenaria Rush-leaved Fescue
What is probably this taxon was scattered on sand at the base of beach cliff. Robust with long rhizomes and large spikelets. SD6
Compt 49: 4779 6740, 4779 6734, 4781 6716 (extending over 6m)
Compt 59: 4778 6604, 4778 6597, 4776 594, 4766 6486
Not recorded (overlooked) in 1993.

Glaucium flavum Yellow Horned Poppy

As with *Crambe maritima*, *Glaucium* has spread and become locally frequent in Compt 49 where the beach had a more gentle profile than in 1993. Many young plants indicated recent colonization. Scarce in Compt 59 but still present on the fine shingle platform at the south end where recorded in 1993. A larger population on the Sizewell Power Station beach not surveyed. SD1.

Compt 49 (684 plants)

4777 6761 – 4778 6755	113 (many young plants)
4777 6753	5
4779 6743	7
4781 6692	25
4782 6678	67
4782 6674	5
4781 6660	18
4782 6659	135 (many young plants)
4782 6654	100 (many young plants)
4781 6646	120 (many young plants)
4781 6638	60 (many young plants)
4781 6633	14
4780 6626	10
4779 6615	5

Compt 59 (16 plants)

4766 6490	1
4766 6475 – 4765 6460	15 shingle platform

Lathyrus japonicus Sea Pea (NSc)

Locally plentiful on semi-fixed dune at the edge of the beach cliff and on the fine shingle platform at the S end of Compt 59. SD1. SD6. SD7. Vulnerable to beach erosion, but this creates suitable habitat.

Compt 49 (5 sites, one with over 50 plants)

4781 6716	Many plants over 30m
4781 6679	24
4782 6678	4
4780 6636	27
4780 6631	1

Compt 59 (8+ sites, four with over 50 plants)

4771 6555	378 plants, low eroded cliff	
4772 6553		
4771 6552		
4771 6550	50	Fine shingle platform 4m at N end, 11m at S end.
4766 6475	70	
4765 6468	2	
4765 6464	87	
4765 6460 – 4764 6457	100s	

Has increased in Compt 59 and spread back into Compt 49 since 1993.

Leymus arenarius Lyme Grass
Scattered along the eroded edge of the dunes, mostly in Compt 59 where recorded at four sites in 1993. Only one fairly large patch. SD6 mobile dune. Vulnerable to erosion.

Compt 49 (one site)

4871 6727

Compt 59 (seven sites)

4779 6600	1 shoot
4775 6575	c. 40 shoots
4771 6555	2 shoots
4772 6553	Many shoots
4770 6543	Extending over 29m on low cliff
4767 6501	On three eroded tussocks fallen from the dune cliff onto the beach
4765 6464	A few shoots on fine shingle platform

Salsola kali Prickly Saltwort
A single plant on sand at base of eroded cliff, Compt 59, 4766 6478. Not recorded on reserve in 1993.

Senecio squalidus Oxford ragwort
Not typically a strandline plant, but a number of plants growing on the fine shingle platform at the southern end of Compt 59, 4764 6457 and 4764 6452, SD1.

Senecio viscosus Sticky groundsel
Growing with *Honckenya peploides* on fine shingle platform at the extreme southern end of Compt 59, 4764 6452, SD1.

4.2 Fixed dune

Campanula rotundifolia Hair-bell
Only recorded in SD10 a short way S of the Sluice, 4772 6590, Compt 59. In 1993 observed in small quantity at three sites in this Compt.

Cladonia portentosa A lichen
Characteristic of dry rather open fixed dune grassland in Compt 59. SD11 and SD12a.

4766 6563	SD12a with *Silene uniflora*
4764 6551	SD12a
4768 6482	SD12a
4762 5465	SD11
4765 6465	SD12a

Corynephorus canescens Grey Hair-grass (Near Threatened)
Restricted to a site at the north end of the beach (National Trust land) and another in Compt 49. Well established but introduced from Norfolk stock after the 1953 floods (Simpson 1982). Both populations occur on desiccated fixed dune on the landward side (SD11a).

National Trust:

4776 67744	c. 330 tussocks immediately S of large Tamarisk *Tamarix gallica* bushes and steps on the landward (west) slope if the eroded shingle bank.

Compt 49:

4778 6704 – 4778 6697	1070 tussocks extending over 76m on landward slope of dry trampled dune just S of gate from path that leads across North Marsh to the dunes from the Minsmere Centre.
4780 6700	Another 50 tussocks beside and on the wide much trampled track on the dune crest close to the previous site.

Vulnerable to trampling and dune erosion. Indeed , a selection of the northern population apparently lost when the sea broke over the shingle ridge in the 2003/04 winter.

Dryopteris dilatata Broad Buckler-fern
Compt 59: Frequent along the outer dune crest, just on the landward (west) side. Also on the inner dune ridge, SD10.
Eg TM 4774 6599
47726585 – 4777 6588
4770 6584
4769 6574
4769 6567
4765 6533 abundant
4765 6528 abundant
4766 6533
4766 6515
4766 6504
4551 6595

Filago minima Small Cudweed
On dry gravely banks with vegetation sparse. With such species as *Aira praecox, Plantago coronopus, Rumex acetosella* and *Sedum anglicum*. SD11b.
Compt 49:
Eg 4776 6631, just N of the Sluice,
Also elsewhere

Jasione montana Sheep's-bit
Scattered along the seaward edge of the dune ridge and mown margins of paths further inland. In dry situations, mostly SD11b but also SD7c

Compt 49:	Compt 59:
4776 6647	4783 6513
4780 6643	4755 6475
4780 6640	
4779 6619	

Polypodium vulgare Polypody
Only recorded at one site near the northern end of the dunes. Compt 49, 4779 6719, but probably on the dune ridge in Compt 59 (see 1993 survey).

Sedum anglicum English Stonecrop
Recorded on a dry gravely bank on the landward slope of the dunes north of the Sluice, Compt 49, 4776 6631 SD11b. Probably in similar situations elsewhere but September not a good time to survey this species.

Silene uniflora Sea Campion
The small patch (1 x 1.5m) refound among *Festuca ovina – Clodania potentosa* between two tracks on the landward side of the main dune ridge, Compt 59, 4766 6565. A more extensive patch was there in 1993. It was not recorded on shingle at the S end of Compt 59 where observed in 1993.

4.3 Heathland

Calluna vulgaris Heather
Very thinly scattered on the inner side of the dunes. In no way can any part of the dunes be identified as more than isolated fragments of Heather-dominated heathland and this negates their proposed open access status.

Compt 49:

4779 6657	SD12a

Compt 59:

4763 6537	SD12a
4758 6502	Small patch of H1 in SD10
4749 6491	SD12a (ram-grazed)
4571 6489	A fairly large patch just N of dense oak/birch woodland and to W of track. Extending in small quantity to E of track.

Deschampsia flexuosa Wavy Hairgrass
Not a typical plant of the East Suffolk heathland. A single small patch adjacent to a large patch of *Calluna* just N of oak.birch woodland at the S end of Compt 59, 4571 6489. H1?

Erica cinerea Bell Heather
Thinly scattered on the dunes and more widespread than *Calluna*. In a few places where associated with *Ulex gallii*, the vegetation could be ascribed to relict H8 *Calluna vulgaris – Ulex gallii* heath but the patches are very limited. Mostly growing in SD12a.

Compt 49	Compt 59
4778 6676	4768 6597
4779 6674	4758 6502 (with *Calluna*)
4776 6656	4755 6475
4775 6653	4752 6470

Ulex gallii Western Gorse
Thinly scatted in Compt 49; one site in Compt 59. flowering, but so was some *U europaeus*.

National Trust:

4779 6739	Shingle ridge

Compt 49:

4779 6731*	With *Ulex europaeus* and *Elytrigia atherica*
4778 6700*	With *Glaux maritima*
4780 6700	With *Ulex europaeus* on dune ridge
4778 6678	Among *U europaeus* and *Rubus fruticosus*
4776 6650	With *Erica cinerea* (H8?)

Compt 59:

4751 6495	SD12a. Ram-grazed area that was oak/birch scrub. One bush.

Four small patches on landward margin of dunes in the northern half of Compt 49 (all 4766) in 1993. described as H8 *Calluna vulgaris – Ulex gallii* heath. At two of the 2004 sites halophytic plants were present, probably as a result of recent sea incursions.

4.4 Halophytes (saltmarsh)

This element of dune vegetation occurred where there was a brackish seepage just north of the Sluice, Compt 49.

Elytrigia atherica Sea Couch

Frequent around two brackish pools just north of the Sluice Compt 49 4775 6635 and 4776 6631 where associated with other halophytes (see below). SM18. Also stands among dry fixed dune grassland SD10 and SD12a.

eg Compt 49: 4779 6740, 4779 6731, 4779 6709, 4777 6624
Compt 59: 4772 6604, 4774 6596, 4772 6567, 6765 6562, 4559 6478, 4795 6466, 4765 6485

Glaux maritima

Frequent with other halophytes around a brackish seepage along the path just north of the Sluice, Compt 49 4775 6639, SM16b. Also on landward side of dune just S of gate onto dunes from the North Marsh path 4780 6607.

Juncus gerardii Saltmarsh Rush

Locally frequent in damp ruts on path just north of the Sluice and in a small area of brackish fen nearby, Compt 49, 4775 6647, 4775 6645 with *Potentilla anserina*, 4775 6639. There is a saline seepage at this point. SM16b and SM18.

Juncus maritimus Sea Rush

Locally frequent around two brackish pools just north of the Sluice, Compt 49 4775 6635, 4776 6631. SM18.

Lotus glaber Narrow-leaved Bird's-foot Trefoil

Along margin of path near brackish pools just north of the Sluice, Compt 49. 4775 6639. SM16b.

Oenanthe lachenalii Parsley Water-dropwort

A few plants with *Phragmites australis* in a small brackish fen north of the Sluice, Compt 49. 4776 6647. S26.

Plantago maritima Sea Plantain
A small quantity with *Agrostis stolonifera, Glaux maritima* and *Juncus gerardii* in a damp rut on path just north of the Sluice. Compt 49 4775 6639. Not recorded in 1993.

Ruppia maritima Beaked Tasselweed
Plentiful in the southern of the two brackish ponds north of the Sluice, Compt 49. 4775 6631. SM2.

Scrub

Cytisus scoparius Broom
Only a few bushes on dunes to N of the Sluice.
National Trust: 4779 6744. Compt 49: 4775 6641

Humulus lupulus
Scrambling over scrub just S of the Sluice. Compt 59. 4772 6606.

Ilex aquifolium Holly
One compact bush on dune ridge, Compt 59 47774 6599.
Not recorded in 1993.

Lonicera periclymenum Honeysuckle
Fairly frequent on the dune ridge, Compt 59 and extending onto the beach where the cliff has been eroded 4768 6522.

Lupinus arboreus
Naturalized on the dunes to the S of the Sluice, Compt 59. Mostly on mobile and semi-fixed dune SD6 and SD7.
4774 6612
4778 6603 (2 bushes)
4772 6604 – landward side near path
4778 6604 (2)
4778 6597 (15)
4777 6594 (2)
4776 6583 (1)

Autumn Ox-eye *Leucanthemella serotina*
A substantial strand in rough vegetation between the Sluice and the bushes to the south 477 660. The South-east European native is clearly introduced and the site is the garden of a former cottage. In full flower (large white daisies) in early October 2004. Determined by Peter Sell, Cambridge University Herbarium.

5 Grazing by sheep

Several fenced plots towards the south end of Compt 59 have been grazed by sheep in recent years. This includes a fairly large plot on the western side of the dunes north of the block of

woodland. Here the grassland had been invaded by Pedunculate Oak *Quercus robur* and Downy Birch *Betula pubescens* scrub. Up to four rams were grazed there for much of two years. They had succeeded in killing off most of the young birch and oak, but the taller birch (over 3m) had largely survived. The vegetation of the area had reverted to SD10. Measurement of vegetation taken on either side the fence line of two enclosures demonstrated a marked effect in its height but little in the species composition (Table 1)

Height (cm)	4759 6474		4769 6422	
	Grazed	Ungrazed	Grazed	Ungrazed
Cladonia portentosa	4,5	13,13	4,4	48,50
Carex arenaria	20,23	55,57	6,9	51,60
Festuca ovina	24,25	45,46	15,16	50,55

	5034 3634	
	Grazed by rams	Ungrazed
Height of ground vegetation (SD10 with *Carex arenaria*)	34, 34, 36, 43, 50	67, 73, 80, 80, 85

6 Changes since 1993

Beach profile

At the north (Dunwich) end of the beach there has been considerable erosion and the narrow dune ridge was over-topped by a surge in the 2003/04 winter which resulted in the deposition of much fine shingle and sand on the ridge.

Over a 500m stretch to the north of Sluice in Compt 49 shingle and sand has built up to give the beach a more gentle graded profile and the dune cliff has disappeared from this sector.

To the south of the Sluice in Compt 59 the dune cliff (up to 5m high) has been cut back by erosion over about a one km stretch.

At the upper part of the beach at the south end of Compt 59 there is platform of fine shingle that extends for 292m along the shore within the reserve. The shingle beach was present in 1993 but the platform had become more defined.

Strandline vegetation

As in 1993, SD1 was best respresented on the fine shingle platform near the southern boundary of the reserve, Compt 59. However, both Sea Kale *Crambe maritima* and Yellow-Horned Poppy *Glucium flavum* are now flourishing on the gentle graded beach to the north of the Sluice Compt 49, from which they were absent in 1993. Sea Holly *Eryngium maritimum* has also spread northwards to just south of the Sluice, in Compt 59. Sea Pea *Lathyrus japonicus* has several strong colonies on the semi-fixed dune (SD7) as well as on the fine shingle platform in Compt 59 and has recolonized Compt 49, where it occurred in the 1970s. Thrift *Armeria maritima*, Prickly Saltwort *Salsola kali* and Rush-leaved fescue *Festuca arenaria* were not recorded on the Minsmere dune in the 1993 survey; Sea Rocket *Cakile maritima* on the other hand appears to have disappeared since 1993.

Dune vegetation

Sand sedge *Carex arenaria* has developed a dense sward and forms the dominant fixed dune vegetation (S10) over most of the outer and higher ridge in Compt 59 to an extent that it probably excludes less competitive species. Nevertheless Broad Buckler-fern *Dryopteris dilatata* is a feature of the landward side of the dune crest.

A Sheep's Fescue *Fetuca ovina* grassland (SD12a) is still a feature of the dry landward slopes of the dunes in Compt 59. Here there are patches of the lichen *Cladonia portentosa* as in 1993. Grey Hair-grass *Corynephorus canescens* contines to be restricted to two areas on desiccated fixed dune (SD11a) on the landward slope in Compt 49. At the northern Dunwich end it is particularly vulnerable to erosion as the colony is situated close to where the 2003/04 winter sea overtopping occurred. The relict population of Sea Campoin *Silene uniflora* persists in Compt 59, but the population has diminished.

Grazing by sheep in Compt 59 had reduced the density and structure of *Carex arenaria* in enclosures to the benefit of *Cladonia portentosa* and *Festuca ovina*. A few rams had effectively checked invasive young Oaks *Quercus robur* and birches *Betula pubescens* in an enclosure, but older, taller birch had survived.

7 Risk assessment for beach and dune vegetation and mitigation of potential damage

Strengthening the sea defences at Minsmere is likely to concentrate on the sector north of the Sluice (Compt 49) and entail use of material from the beach.

Vulnerable are:

- The newly established population of Sea Kale *Crambe maritima* and Yellow-horned Poppy *Glaucum flavum* on the beach (4777 6761 – 4776 6675).

- A strong colony of Sea Pea *Lathynus japonicus* (NSc) over a 30m stretch of outer dune just north of the gate onto the dunes from the North Marsh path (4781 6716) plus at least four small colonies.

- Two populations of Grey Hair-grass *Corynephorus canescens* (Near Threatened) on the landward slope of the dunes; one at the Dunwich end (4779 6744) and the other just south of the gate onto the dunes from the North Marsh path (4778 6704 – 4778 6697 and 4780 6700).

  The Sea Kale and Yellow-Horned Poppy are short-lived plants and provided a source of seed survives in the vicinity would recolonise the beach naturally, if it is not steeply inclined and wide enough to avoid much erosion.

  The Sea Pea grows on the semi-fixed dune at the top of the beach. It too would probably re-establish naturally from a local seed source, but less readily than the previous two species. Recolonisation could be aided by sowing seed from local sources (ie to the south of the Sluice).

  The Grey Hair-grass at Minsmere was apparently introduced from Norfolk stock in 1953, but the grass occurs naturally along the Covehithe coast to the north. It is a perennial and does not spread readily. Once a colony is destroyed it would have to be

reintroduced by planting. Every effort should be made to protect the sites from damage either from erosion or engineering works. While sea defences are being raised it may be necessary to temporarily remove the dune turf where it is growing.

- South of the Sluice (Compt 59) the dunes are both higher and wider. Improving sea defences is much less urgent. Every effort should be made to protect the fine shingle platform at the upper part of the beach at the southern (Sizewell Power Station) end of the reserve (4766 6475 – 4764 6460). Here there is a good strandline flora and strong population of both Sea Kale and Sea Pea as well as Sea Holly *Eryngium maritimum*. This site provides a source of seed for recolonization on neighbouring shores.

8 Acknowledgements

I am grateful for help given me by Ian Hawkins, Senior Warden at Minsmere Reserve

9 References

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Assessing the Impact of the Tidal Surge of 5/6 December 2013 on the Shore Profile and Flora of RSPB Nature Reserves on the Norfolk and Suffolk coast

C James Cadbury



Sea Bindweed

January 2015
RSPB Reserves Ecology, Sandy, Bedfordshire SG19 2DL

Assessing the impact of the tidal surge of 5/6 December 2013 on the shore profile and flora of RSPB nature reserves on the Norfolk and Suffolk coast

The situation

A major surge of seawater (3.1 m AODN) in the North Sea hit the Norfolk and Suffolk coast overnight on 5/6 December 2013. Immediately prior to the surge the wind was northerly but only light and the tide was nothing exceptional (high tide at King's Lynn at 1915 hrs on 5 December was 7.1 m). A surge had been forecast and on the 5th the sea retreated alarmingly at such places as Eccles on the north-east Norfolk coast. Overnight a massive wall of water, some two metres above the normal high tide level, swept onto the shore, overtopping the dunes and sea defences in a number of places. Havergate Island on the River Ore, East Suffolk, was almost completely inundated and the surge continued up the river to Snape. Following the surge the sea retreated fairly rapidly, unlike the situation in 1953 when a strong north-east wind coupled with a high tide held up the flood water for several days, causing widespread devastation.

The cause of this tsunami-like event was high atmospheric pressure in the northern part of the North Sea that drove a huge volume of water southwards into a low pressure area off the East Anglian coast.

Project objectives

Concentrating on the RSPB's reserves on the Norfolk and Suffolk coast these were:

a) To examine the shore profile to assess what changes may have resulted from the tidal surge.
b) To record the coastal flora in areas affected by the surge and where possible make comparisons with information obtained earlier.

Methods

Much of the recording effort was along transects from about the mid-high tide level, across the beach, over the dunes to the grassland or saltmarsh inland. The different zones, whether it be substrate (shingle, sand or mud) or vegetation on pioneer, building or stable, mature dunes, dry grassland or saltmarsh were recorded and measured in paces. The transects were about 10 paces wide. All species of vascular plants were noted and their frequency recorded. Particular attention was given to those species associated with the more mobile, less stable substrates on the seaward side of the shore. At Titchwell, however, attention was given to the high saltmarsh flora of depressions landward of the main dunes since these supported communities special to north Norfolk that have affinities with those of more southerly coasts of Europe, including the Mediterranean. Previous surveys of the more notable plants enabled some comparisons of distribution and populations.

Locations

The survey covered 6 RSPB reserves.

a) **Havergate Island** (East Suffolk, vc 27) where there are no dunes, but a stable shingle ridge bordered by a steep shingle beach on the south-east side and relatively high saltmarsh to the north-west, west of the accommodation huts, and high saltmarsh beside the River Ore to the north-east, outside the seawall protecting the lagoons. At the north end of the island the seawall has been deliberately breached.

b) **North Warren** (East Suffolk) where a shingle beach extends from the car park at the north end of Aldeburgh to the south end of Thorpeness (The Haven) to the east of the coast road.

c) **Minsmere** (East Suffolk), that extends north from the northern boundary of Sizewell Power Station at the southern end of the reserve, past the New Cut Sluice to the Dunwich Heath scarp (National Trust). There is a well defined dune ridge along this stretch of coast.

d) **Dingle** (East Suffolk) extends from the car park at the north end of Dunwich to the path that leads from the beach to Dingle Great Hill. The 2014 survey only continued for one km north of the car park to a fence across the beach. There is a shingle beach that narrows northwards and is severely truncated by erosion at its northern end. A reedbed borders the inland, west side of the shore.

e) **Titchwell** (West Norfolk, vc 28). The survey covered the stretch of coast from Brancaster golf club, past the seaward end of the path from the RSPB reserve Centre to the beach, to Thornham Point. There is a sand dune all along, backed by saltmarsh and the reserve's Tidal Lagoon.

f) **Snettisham Pits** (West Norfolk). This on the east side of the Wash. There is a narrow shingle beach on the seaward side of the Pits and north of the bungalows there is another shingle beach (The Scalp) south of the Yacht Club.

More details of the transects and grid references of sites are presented in a separate hand-written report.


TITCHWELL
SNETTISHAM
DINGLE
MINSMERE
NORTH WARREN
HAVERGATE

Havergate Island

2 visits 13 May & 18 September, 2014

One transect (13.5) from beach on SE shore (TM 40804686) to river edge of saltmarsh on NW shore, between Doveys Lagoon and the accommodation huts, 7 vegetation zones identified and recorded. SE shore from Doveys Lagoon – opposite Main Area covered. Second visit to record *Salicornia* spp., *Sarcocornia perennis* and *Spartina maritima*. Previous botanical information from 1971, including 2013.

The surge inundated most of the island, destroying the Gullery Lagoon Hide (picked up and turned 45° and Main Area Reception Centre. Main Area Hide badly damaged, having been moved several metres. Accommodation huts flooded to depth of 45 cm. Subsequently Doveys Lagoon bank lowered to allow flood water to be discharged.

SE shore of the island eroded to the effect that the shingle beach between the Huts and Doveys has become increasingly steep.

Flora

Zones along transect.

A 5 m. Shingle beach and eroded cliff (SE shore). Scattered Sea Beet *Beta maritima*, Sea Campion *Silene uniflora* and Sea Mayweed *Tripleurospermum maritimum*. 2 small patches of Sea Kale *Crambe maritima.*

B 13-23 m. Vegetated shingle ridge with a high density of nesting Herring and Lesser Black-backed Gulls. Sea Campion (abundant), Bitter Stonecrop *Sedium acre* (frequent). Early Scurvy-grass *Cochlearia danica* (F), Squirrel-tail Fescue *Vulpia bromoides* (F), Sheep's Sorrel *Rumex acetosella* and Sea Mouse-ear *Cerastium diffusum*. Ridge has become increasingly vegetated as the result of nutrient enrichment by the gulls.

C 7 m. NW slope of ridge: Red Fescue (abundant). Early Scurvy-grass (A), Sheep's Sorrel, Sea Mouse-ear and Sea Beet.

D 2-4 m. Lower slope of ridge bordering track. Sea Wormwood (abundant), Saltmarsh Couch *Elymus atherica* (A), Hoary Cress *Cardaria draba* (locally A) – probably recent coloniser.

2 m Track. Little vegetation in May but by September One-flowered Glasswort *Salicornia pusilla* (NSc) and Annual Seablite *Suaeda maritima* both abundant.

E F G 69-78 m. Saltmarsh (NW side), Sea Purslane Atriplex portulacoides (abundant), Sea Aster *Aster tripoleum* (A upper saltmarsh), Common Sea-lavender *Limonium vulgare* (F), Long-leaved Scurvy-grass *Cochlearia anglica*, Sea Arrow-grass *Triglochin maritimum*, Thrift Armeria maritima (A mid-saltmarsh), Small Cord-grass *Spartina maritima* and Perennial Glasswort *Sarcocornia perennis* (both around shallow pools, upper saltmarsh).

Detrimental effects of the surge:

Yellow Vetch *Vicia lutea* (NSc) recorded as locally frequent on stabilised shingle ridge SE of Doveys Lagoon in 9 years 1971-2013. Not refound in 2014, either in May or September. Brown Hares graze it but their population reduced after 2013. A candidate for local extinction as result of the surge.

Other species that have apparently suffered are Rock Sea Lavender *Limonium binervosum* ssp. *saxonicum* in shingle extraction pits near the Huts (decrease), and disappearance from 2 sites on shingle ridge between Huts and Doveys.

Bird's-foot Clover *Trifolium ornithopodioides* and Suffocated Clover *T. suffocatum* (NSc) – single small populations recorded in 2013 but not refound in May 2014.

Gorse *Ulex europaeus*: a large patch at N end of the Main Area Meadow all dead, but bushes on ridge E of Doveys Lagoon living. Those N of Huts showed signs of dying.

Disappeared before the surge:
Sea Pea *Lathynus japonicus* (NSc): locally abundant on shingle ridge between the Huts and Doveys Lagoon in 1971 but much reduced after 1976 drought. Not seen in 2011 and 2014. Probably still on Orefordness where it was abundant.

Slender Hare's-ear *Bupleurum tenuissimum* (NSc, Vulnerable): appeared locally in abundance following substrate disturbance in the Meadow area S of the Main Lagoon in the 1970s and 1997 but not seen subsequently. Cannot compete with coarse grasses.

Rock Samphire *Crithmium maritimum*: a large plant seen on the shingle bank SW of the Huts in 2008 but not recorded subsequently. Probably a temporary colonist where it is at its northern limit on east coast of Britain.

Species unlikely to have been affected by the surge:
Perennial Glasswort *Sarcocornia perennis* (NSc), 5 species of Annual Glassworts *Salicornia* spp. including Yellow Glasswort *S. fragilis* (NSc), though *S. pusilla* (NSc) was particularly widespread and frequent in 2014.

Small Cord-grass *Spartina maritima* (NSc, Endangered): Havergate is an important site for this species, with good populations associated with shallow pools on the upper saltmarsh.

Sea Barley *Hordeum marinum* (NSc, Vulnerable): has been abundant on Belper's Lagoon in 2001, 2011 and 2014 (local). This annual grass responds to substrate disturbance in brackish sites.

Borrer's Saltmarsh-grass *Puccinellia fasciculata*: recorded from several brackish muddy sites in 2001 and 2011 but not seen in 2014.

Curved Hard-grass *Parapholis incurva* (NSc): locally frequent on sparsely vegetated damp ground (eg tracks) in 2011 but not in 2014. The dry summer of 2014 may have affected both grasses.

Tassel-weed *Ruppia maritima*: abundant in the brackish Meadow Ditch in 2011 and 2014.

Sea Kale *Crambe maritima*: has been confined to the shingle bank on SE side of the isthmus between the Huts and Dovey's Lagoon where a small population of 7-18 plants has been recorded 2001-2014. It was considered to be a new coloniser in 2001 from Orfordness. There were 14 plants in 2014. Periodically plants are washed away and replaced by new ones (from seed) as in 2008. It probably benefited from the turn-over of shingle in the surge.

North Warren, East Suffolk (vc 25)

Shore surveyed on 23 July 2014 from between the car park at the N end of Aldeburgh to the south end of Thorpeness (The Haven). TM 46895815-47095899, a distance of c1.5 km.

A steep unstable shingle slope with little vegetation rises from the broad (20-30 m) shingle beach. The shingle beach crest is 3-12 m in width. The well vegetated landward slope is composed largely of shingle that had been swept over the crest by the surge. Inland of the main ridge is a broad expanse of grassy vegetation (78-108 m wide). At the N (Thorpeness) end the shingle had been carried landward by up to 84 m from the beach crest. Here the beach crest had been lowered and it appeared that the surge had carried material up to the fence bordering the garden of Haven House.

Sea Pea *Lathyrus japonicus* (NSc) is a feature of the beach between Aldeburgh and Thorpeness, particularly at the southern (Aldeburgh) end. In 1998 and 2000 there was a large population that extended over a strip of about 1 km x up to 16 m wide where it formed up to 50% cover on the southern part of the beach south of Sluice Cottage. It benefited from the roping off of part of the beach crest to protect a Little Tern colony and prevent trampling. In 2014, following the surge, *Lathynus* remained abundant at the south end. Its distribution extended onto the landward slope up to 27 m from the beach crest. The surge had clearly pushed the shingle over the ridge landward taking with it certain traditional beach plants including the *Lathyrus*.

Sea Kale *Crambe maritima* was locally abundant on the beach crest in 2010, particularly in the roped off area at the more southerly end. In 2014 it was abundant at both the south and north ends of the beach crest. There were many small plants on the steep shingle slope at the upper part of the beach where recent colonisation may have taken place. In one place 6 plants were growing up to 34 m inland of the ridge where the surge had carried shingle 84 m landward.

Yellow Horned-poppy *Glaucium flavum* was occasional along the beach ridge in 2014 but at one place it was growing with *Crambe* 34 m inland of the ridge. It was frequent along the fence of Haven House garden where seed appeard to have been transported by the surge and probably earlier as several plants were in flower.

Sea Holly *Eryngium maritimum* was only occasional on the landward slope of the beach crest towards the north end and up to 50 m landward from the crest in 2014.

Sea Spurge *Euphorbia paralias*: only about 15 plants on the beach crest and adjacent landward slope at the north (Thorpeness) end in 2014.

Dune Fescue *Vulpia fasciculate* (Nsc) was recorded at 3 sites on the south side of Thorpeness, including a car park, in 2010. It was not looked for in 2014.

Apart from shifting shingle onto the landward side of the beach ridge and lowering the ridge at the north end, the December surge had no major effects on the flora of Aldeburgh to Thorpeness shore.

The verge of the coast road (Thorpe Road) between Aldeburgh and Thorpeness has an interesting flora, particularly to the north of Sluice Cottage on the east side, and was not affected by the surge. Dittander *Lepidium latifolium* was well established there by 2003 and extended over 150 m south of Sluice Cottage. Both White Stonecrop *Sedum album* and Reflexed Shorecrop *S. rupestre* are naturalised. In 2003 there were hundreds of plants of Rough Dog's-tail *Cynosurus cristatus* on the east bank of the road but in 2010 there were only a few and none in 2014 by which time much of the bank had become vegetated over. In 2002 Sand Catchfly *Silene conica* (NSc, Vulnerable) was present at 3 sites on the east bank. It reappeared in 2012 but none was seen in 2014 following excavations to discourage car parking. Some scarification is, however, probably necessary for the existence of the annual. In May 2013 Bulbous Meadow-grass *Poa bulbosa* (NSc as a native) was discovered at 7 sites on sandy gravel disturbed groud in parking places and entrances to tracks on the east side of the coast road. This early flowering grass was not looked for in 2014 and may have been overlooked in previous years.

Minsmere East Suffolk (vc 25)

Fortunately the reserve survived the surge largely unscathed. There was almost no overtopping of saltwater into the freshwater habitats.

The north end of the shore from Dunwich Heath/cliffs to the New Cut Sluice (TM 477675-473661) surveyed on 2 July 2014. Eight transects from the shore-line to the bank bordering the North Marsh and East Scrape (Compt 49) carried out. A second visit was undertaken on 24 September to survey the shore from the Sluice to the concrete blocks that mark the boundary between the southern end of the reserve and the Sizewell Power Station land (473661 and 476646, Compt 59). No transects taken but the height of the dune cliff measured.

Between the sea and the dune ridge in Compt 49 there was a broad shingle beach, 32-51 m wide that was almost devoid of vegetation except in Transects 7 and 8 which had typical early phase beach colonisers. The dune ridge on its seaward side was eroded, forming a cliff, particularly south of the Sluice where it varied in height from 0.5-3.0 m. The landward slope of the dune ridge was well vegetated and to the north of the Sluice sloped down to a well-used path, though in Transects 2 and 3 the path had been obliterated by shingle that had been washed over the ridge by the surge. To the south of the Sluice the landward slope was broad and stabilised by dry grass and sand sedge.

Sea Kale *Crambe maritima*
In 1993 225 plants (56% small, 16% medium and 28% large) were counted on the beach in Compt 59, south of the Sluice. In 2004 there were 267 plants in Compt 59 and there had been a spread in Compt 49 to the north of the Sluice where there were 132 plants. In 2011 at the Dunwich (north) end of the beach where the sea had breached the ridge in the previous winter there was only a single small plant, but there were many large plants on the beach opposite the Public Hide in Compt 49.

In 2013, apart from a cluster of seedlings well up the beach among Marram, there were 11 plants in Compt 49 and hundreds to the south of the Sluice in Compt 59.

In 2014, 4 plants (2 large) at the north (Dunwich) end of Compt 49 and it was abundant on the upper beach towards the southern end of this compt. South of the Sluice in Compt 59, 310 plants were counted, mostly at the North end. There were 8 plants 10 m inland of the low cliff where the surge had carried shingle over the ridge.

Yellow Horned-Poppy *Glaucium flavum*
In 1993 there were a few small plants at the Dunwich end of Compt 49. In Compt 59 there were 57 seedlings on the strandline and 2 flowering plants at the southern boundary.

By 2004, where the beach had developed a more gentle profile in Compt 49, 684 plants were counted, including many young ones, indicating recent colonisation. In 2011 where the sea had breached the ridge at the Dunwich end of Compt 49 there were 49 flowering plants and 50 seedlings.

In 2014 there were 14 flowering and 2 young plants on the ridge at the north end of Compt 49 and 107 with only 13 flowering in Compt 59.

Babington's Orache *Atriplex globriuscula*
In 2014, frequent on the beach shingle in Compt 59. Not recorded in Compt 49. Unrecorded or overlooked in previous years.

Sea Pea *Lathyrus japonicus* (NSc)
From the 1970s to 2004 locally plentiful in Compt 49 north of the Sluice and in Compt 59 to the south where 500 plants recorded on a fine shingle platform at the south end of the reserve. An increase noted between 1993 and 2004. Not recorded in 2011 or 2012 from Compt 49, and rare there in 2013. In 2014 just 17 plants recorded in the southern part of Compt 49 and at 3 sites in Compt 59. The shingle platform where abundant in 2004 had been washed away.

Sea Holly *Eryngium maritimum*
In 1993, 12 plants recorded at the south end of Compt 59.

In 2004, there was a single plant in Compt 49 and 59 plants in Compt 59 including 26 on the fine shingle platform at the south end of the reserve.

In 2014 it was not recorded in Compt 49; there was a total of 24 plants at 2 sites in Compt 59 but the shingle platform had been washed away.

Other strand-line species that were recorded rarely and sporadically.

Prickly Saltwort *Salsola kali* (Vulnerable) was only recorded as a single plant in 2004 in Compt 59.

Sea Beet *Beta maritima*: in 2004 there was a single plant at the north end and 7 plants on the fine shingle platform in Compt 59. This latter site had been washed away by 2014.

Rock Samphire *Crithmum maritimum*: in 1993 there was one plant south of the reserve's boundary fence (Sizewell) and in 2004 3 plants appeared south of the Sluice in Compt 59 but was not seen in 2014. On the east coast of 'England' this species does not extend further north than East Suffolk.

Sea Rocket *Cakile maritima*: appeared on the strandline in Compt 59 in 1993 but has not been recorded since.

Sea Sandwort *Honckenya peploides*: a strandline plant that seems to be restricted to a few sites in Compt 59. In 1993 in 2 areas towards the southern end, NVC SD2. 2004 on the fine shingle platform at the extreme south end of the Compt, NVC SD1.

Sticky Groundsel *Senecio viscosus* and Oxford Ragwort *Senecio squalidus* were both growing on the fine shingle platform at the south end of Compt 59 in 2004.

Sea Bindweed *Calystegia soldanella*
1993 & 1994: scattered along the eroded edge of the dune above the beach in Compts 49 and 59. In 2014 frequent on shingle and sand on upper beach and low cliff at the southern end of Compt 49 and on steep banks just south of the Sluice in Compt 59.

Rush-leaved Fescue *Festuca arenaria* (*F. rubra* ssp oraria/*arenaria*)
2004: scattered on semi-mobile sand at the base of the beach cliff in Compt 49 and 59. In 2014 recorded in similar situations in Compt 59. Not recorded (overlooked?) in 1993.

Lyme-grass *Leymus arenarius*
1993, 2004 & 2014: scattered along the eroded edge of the dunes, mostly in Compt 59.

Hybrid Sea Couch *Elymus x obtusiusculus* (*E atherica* x *E farctus*)
2013: abundant on dune ridge south of gate onto the shore.
2014: not recorded in any of the 8 transects in Compt 49 north of the Sluice but a few clumps on dune ridge in Compt 59, including where shingle had been washed over the ridge by the surge.

Grey Hair-grass *Corynephorus canescens* (Rare native, Near Threatened)
Apparently introduced to Minsmere with Marram from Norfolk after the 1953 floods (Simpson 1982) but native in East Suffolk at sites south from Kessingland. Restricted at Minsmere to Compt 49 north of the New Cut Sluice.

1993, 1997, 2003 and 2004: a population of up to 330 tussocks at north end of the dunes (National Trust land). Another large colony (1070 tussocks extending over 76 m in 2004) south of gate from track onto the shore. A third colony (50 tussocks in 2004) to the south.

In 2011, following the sea breaking over the shingle ridge at the North end in November 2006 and 2007 and washing southward down the path none seen at any of the 3 sites.

2012-2014: reappeared at the site south of the gate (TM 47786604-47786697) with 92 tussocks in 2012, 90 in 2013 but only 42 in 2014. *Corynephorus* requires mobile sand. This site on a dry landward-facing bank has become increasingly overgrown with *Carex arenaria*, *Agrostis capillaris* and

Festuca rubra. *Corynephorus* also vulnerable to heavy trampling and seems to be susceptible to saline water.

Thrift *Armeria maritima*
2004: s small patch on the landward slope at the North (Dunwich) end of the dunes. 2014, 27 clumps in this area which is subject to the eroding effect of sea incursions, including the surge. Two clumps further south in Compt 49.

The following plants occur on the landward side of the dunes and are unlikely to have been affected by the 2013 surge.

Sheep's-bit *Jasione montana*
1993, 2004 and 2014: scattered along the dune ridge where the vegetation is short at the southern end of Compts 49 (2 transects in 2014) and 59 (2 sites in 2014).

Bell Heather *Erica cinerea*
1993, 2004 and 2014: thinly scattered on the dunes in both Compt 49 (one transect in 2014) and 59 (one patch 1.5 x 1.5 m in 2014).

Ling *Calluna vulgaris*
1993: 2 clumps west of the track at the south end of Compt 59.

2004: thinly scattered on the landward side of the dunes NVC SD12a, one patch in Compt 49, 4 sites in Compt 59 including a fairly large patch just north of dense woodland and west of track at the south end.

2014: not seen but main site in Compt 59 not covered.

English Stonecrop *Sedum anglicum*
This species has a largely coastal distribution in East Suffolk, the only area where it is frequent on the east coast of Britain.

1993, 1994, and 2014: locally frequent in dry, sparsely vegetated locations on the landward side of the dunes in both Compts 49 and 59.

Harebell *Campanula rotundifolia*
1993: in small quantity at 3 sites on vegetated stable dune in Compt 59. in 2004 only recorded at one site a short way south of the Sluice, Compt 59. In 2014 at 3 sites in this Compt, including on a dune cliff.

Small Cudweed *Filago minima*
1993: on a dry gravelly east facing bank just north of the Sluice, Compt 49.

Sea Campion *Silene uniflora*
1993: in small quantity among *Festuca ovina* and *Cladonia portentosa* between two tracks on the landward side of the dune's Compt 59. 2004: only one small patch (1 x 1.5 m) there NVC 12a. Not seen at this site in 2014.

Wavy Hair-grass *Deschampsia flexuosa*: 1993 and 2004, a small patch adjacent to a large patch of *Calluna* just north of oak-birch woodland at the south end of Compt 59. Site not visited in 2014. An acid, grassland plant.

Polypody *Polypodium vulgare*
1993: on west side of dune ridge, Compt 59. NVC SD10b.

2004: one site at north end of dunes, Compt 49 but probably unrecorded Compt 59.

2014: a small patch on dune, Compt 59. Not in any of the 8 transects in Compt 49.

Broad Buckler-fern *Dryopteris dilatata*
1993 & 2004: locally frequent on west (landward) side of the dune ridge in Compt 59. NVC SD6f & g. 2014: apparently much less frequent: in small quantity at 3 sites (one with 13 plants and another with 7).

Lichen *Cladonia portentosa*
1993 & 2004: on dry gravelly are just north of the Sluice (Compt 49) with *Filago minima*, *Rumex acetosella*, *Aira praecox*, *Plantago coronopus* and *Sedum anglicum*. A feature of dry more level ground beside the main path on the landward side of the dunes in Compt 59. NVC SD11 and 12a.

2014: much less frequent in Compt 59.

Shrubs and dunes:
Western Gorse *Ulex gallii*: essentially a plant of old heath in the eastern part of East Suffolk. Has a largely western distribution in Britain, and is absent from more central parts.

1993, 2004 and 2014: thinly scattered on the landward slope of the dunes in Compt 49. In 2014 some large patches which showed no signs of dying.

Common Gorse *Ulex europaeus*: 2014: 9 bushes on the dunes in Compt 59; one 15% dead, one 10% dead, 2 5% dead, 3 3% dead and 2 1% dead (24.9). 5 bushes had considerable patches in full flower on 24 September suggesting hybridisation with *U. gallii* but shoots resembled those of *U. europaeus*.

Downy Birch *Betula pubescens*: 2014: 3 compact bushes in Compt 59; one 33% dead, one 22% dead and a third 10% dead (24.9).

Holly *Ilex aquifolium*: 2004 & 2014: one compact bush on dunes at north end of Compt 59. In 2014 shoots on the seaward side dead, but healthy on north (landward) side.

Elder *Sambucus nigra*: 1993: scrub at north end of Compt 49 and north end of Compt 59. 2014: 24 bushes in Compt 59, mostly at the north end; all dead except one sprouting at the base.

Broom *Cytisus scoparius*: 2004: only a few bushes at 2 sites on dunes to the north of the Sluice, Compt 49. 2014 4 bushes on a dry bank just north of the Sluice, Compt 49.

Tree Lupin *Lupinus arboreus* (naturalised): 1993: in some quantity just south of the Sluice, Compt 59. 2004: at least 24 bushes at 7 sites, mostly on mobile or semi-fixed dune in Compt 59. 2014: 13 bushes and 9 seedlings at the north end of Compt 59 – 2 dead, 3 part dead and 5 thriving bushes as were the seedlings. Two live and 2 dead on the upper beach on the rest of the dunes in Compt 59.

Honeysuckle *Lonicera periclymenum*: 2014: Compt 59, extending onto the beach where the cliff had been eroded. Abundant and flowering on the vegetated dune. Probably has increased.

Hawthorn *Crataegus monogyna*: 1993: one live bush at the south half of Compt 59. 2014: 5 dead bushes, possibly *Crataegus*, on beach cliff and another 10 m landward of the cliff, Compt 59.

Brackish seepage influence at south end of Compt 49, just north of the Sluice.

1993, 2004, & 2014: saline runnel along path at the margin of 2 brackish pools have halophytes - *Juncus gerardii*, *J. maritimus*, *Aster tripoleum*. *Glaux maritima*, *Spergularia marina* (1993), *Plantago maritima* (2004) and *Lotus glaber* (2004). Tassel-weed *Ruppia maritima* in the brackish pools.

Parsley-leaved Water-dropwort *Oenanthe lachenalii*.
1993, 2004, 2014: a few plants among *Phragmites* in a damp hollow near the path just north of the Sluice. 8 plants flowering in 2014 but the site has become drier.

Adder's-tongue *Ophioglossum vulgatum*: in the same ready hollow, mid 1970s, 1993 & 2004. Not seen 2014.

Effects of the December 2013 surge

Sea Kale: little change in population, some change in distribution as the result of shingle being rearranged.

Yellow Horned poppy: probably temporary decline at north end of beach.

Sea Pea: minor recovery in Compt 49 but a large population on the shingle platform at the south end of Compt 59 washed away.

Babington's Orache: may have benefited on beach in Compt 59 but not recorded or overlooked in 1993 and 2004.

Sea Holly: little change in the population but one of the main populations on the shingle platform at the south end of compt 59 washed away.

Sea Beet: a small population in 2004; not recorded in 2014.

Sea Bindweed: little change.

Hybrid Sea-couch: may have suffered a considerable decrease between 2013 and 2014 as the result of erosion.

Grey Hair-grass: virtually disappeared at the 3 sites in Compt 49 following sea breaching the ridge at the north end in 2011. Reappearance at one site 2012-2014 but decline in population in successive years.

Elder: the bushes just south of the Sluice, Compt 59, all dead except one in 2014. Their condition in previous years not known. Probably not a surge effect.

Common Gorse: bushes on the dunes in Compt 59 showed signs of dying, but this may have resulted on storm exposure.

Former comprehensive surveys of the beach and dunes
1993: Cadbury, C. J. (1996) Vegetation survey of the Coastal Grazing Marshes and Dunes at Minsmere RSPB Nature Reserve, 1993. Unpubl. rept. Reserves Ecology. RSPB, Sandy.

2004: Cadbury, C. J. (2004). Repeat vegetation survey of the Beach and Dunes at Minsmere RSPB Nature Reserve, Suffolk, 2004. Unpubl. rept. Reserves Ecology, RSPB, Sandy.

Dingle, north of Dunwich, East Suffolk (vc 25)

Surveyed from the car park at the north end of Dunwich (TM 47927086) to the electric fence across an extensive spread of shingle at the north end (48237167). Total distance 1.3 km, 11 June 2014. With James McGill 4 transects from the tide line across the beach and ridge to the reedbed at the edge of the marsh.

	Transect	**2011**	**2014**
Beach crest width	3	14-18 m	11 m
	4	4 m	1 m

Zone A. A wide shingle beach, including the seaward slope of the beach ridge and much of the beach crest. 43 m wide at the southern end (Transect 1) narrowing to 18 m (Transect 4). Almost unvegetated except for a few plants of Yellow Horned-poppy *Glaucium flavum* on the beach crest.

Zone B. Vegetated landward slope of beach ridges, 11-19 m wide but beach crest only 1 m wide in Transect 4.

Vegetation largely composed of Sea Campion *Silene uniflora*, Cat's-ear *Hypochaeris radicata*, Sea Pea *Lathyrus japonicus* in Transect 1 (abundant) and 2 (frequent), Sea Beet *Beta maritima*, Rest Harrow *Ononis repens* and Yellow-Horned-poppy (occasional in 3 transects).

Zone C. Litter heaps (presumably deposited by the surge) on the landward side of the beach ridge, 15-26 m wide.

Vegetation mostly Sea Campion, Sea Mayweed *Tripleurospermum maritimum*, Spear-leaved Orache *Atriplex protrata* and Sea Beet.

Zone D. Dry grassland on shingle 13-35 m wide.

Vegetation mostly Red Fescue *Festuca rubra*, Sea Campion, Buck's-horn Plantain *Plantago coronopus* and Sea Beet.

Zone E. Untrampled, ungrazed brackish damp grassland at the edge of reedbed, 5-9 m wide.

Red Fescue (abundant), Saltmarsh Couch-grass *Elymus atherica* (frequent), Sea Rush *Juncus maritimus* (3 transects, frequent in one), Silverweed *Potentilla anserina* (abundant in 1 transect). Divided Sedge *Carex divisa* (localy frequent).

More notable species:

Sea Kale *Crambe maritima*: in 2011 23 plants (15 large, 7 medium, 1 small) at southern (Dunwich) end of beach where shingle ridge still wide. 23 plants (1 large, 5 medium and 17 small where ridge narrowed and lowered and the sea had broken over the top further north. The higher proportion of small plants indicative of recent colonisation.

In 2014: a total of 50 plants (11 large, flowering, 7 medium and 32 small, non-flowering at 8 sites). 78% of the small plants were on the seaward slope of the beach crest; 64% of the large plants were on the beach crest or landward slope.

Sea Pea *Lathyrus japonicus* (NSc). In 2011, 4 large patches (>2 m in diam), 4 medium patches (0.5-2.0 m diam) and 72 small patches (<0.5 m diam). All 4 large patches and 3 of the medium ones were on the main uneroded ridge at the Dunwich end (TM 479710-483720). In 2014, 8 large patches, 5 medium patches and 49 small patches, all on the landward side of the beach crest where shingle pushed by the surge from the beach crest inland. 12 of the 15 sites were at the southern (Dunwich) end of the beach, 480710-48712. Towards the north end, 48027124, 20 small plants were growing on shingle washed almost to the reedbed.

Yellow Horned-poppy *Glaucium flavum*.

2011: occasional on both the eroded beach with sparse vegetation and on the landward side of

the main ridge.

2014: only recorded at the Dunwich car park end of the beach crest.

Sea Fern-grass *Catapodium marinum*
2014: at two sites just on the landward side of the beach crest, at one locally frequent.

Common Gorse *Ulex europaea*
2014: 12 bushes on the landward side of the beach crest. Of these 6 were dead, 2 had dead top shoots, and one plus 3 small bushes were alive.

Species unlikely to have been affected by the surge:
Borrer's Saltmarsh-grass *Puccinellia fasciculata* (NSc, Vulnerable).
2011 and 2014: locally frequent on a damp bare path on the seaward side of the reedbed. Extended over 29 m in 2014.

Curved Hard-grass *Parapholis incurva* (NSc)
2011 and 2014: locally frequent on a damp gravelly path on the seaward side of the reedbed with *Puccinellia fasciculata*.

Divided Sedge *Carex divisa* (NSc, Vulnerable)
2014: at 3 sites in damp grassland on the seaward margin of the reedbed (Zone E). Locally frequent at one of the sites.

Marsh Sowthistle *Sonchus palustris* (NSc)
2014: scattered small plants at the edge of the reedbed, N of the Dunwich car park. With *Carex divisa* and Parsley-leaved Water-dropwort *Oenanthe lachenalii*.

Tasselweed *Ruppia maritima*/*R. cirrhosa* (NSc)
2011: abundant in channel by path from Dingle Great Hill to beach (TM 488731). Identified (on 28.6) as *R. maritima* but non-flowering.

2014: abundant in brackish pool (482717). Identified (11.6) by spirally twisted peduncles as R. cirrhosa.

Saltmarsh: TM 482717. A small area near the brackish pool (above) with *Atriplex portulacoides*, *Aster tripoleum*, *Carex distans*. *Elymus atherica*, *Glaux maritima*, *Juncus gerardii*, *J. maritimus*, Puccinellia maritima, *Spergularia media*, *Salicornia ramosissima* and *Triglochin maritimus*.

Titchwell, West Norfolk (vc 28)

This stretch of coast was subdivided into three sectors and surveyed over three days.

a) West from Brancaster golf club-house TF 770452 to the tidal creek 7605 4469, a distance of c1 km. Six transects surveyed from high tide mark across the dune to the saltmarsh on 6 August 2014. Included a dry saltmarsh depression in Transect 2.

b) East from the seaward end of the path from the RSPB's Titchwell Reserve Centre, 750451, to the first tidal creek 760448, a distance of c1 km. Four transects surveyed across the shore and dune to the Tidal Lagoon, 29 July 2014.

c) West from the seaward end of the path to Thornham Point, 742452, a distance of c1 km. Four transects surveyed from the high tide mark across the dunes to the saltmarsh, 8 July 2014. In addition 4 transects were covered across the extensive saline depression landward of the dunes.

a) Titchwell (Brancaster sector)

Zone A: a broad unvegetated sandy beach (or creek margin Transect 5), 103 m in the east (Transect 1) decreasing to 20 m at the tidal creek (Transect 5) where there was c0.5 m high mud cliff.

Zone B: a narrow embryonic dune, 4-21 m wide, where both Prickly Saltwort *Salsola kali* (Vulnerable) and Frosted Orache *Atriplex lacireata* were frequent or abundant; Sea Couch-grass *Elymus farctus* was plentiful.

Zone C: a wider sandy fare-dune, 13-24 m (absent from Transect 6) where Prickly Saltwort and Frosted Orache featured prominently along with Sea Sandwort *Honckenya peploides*, Sea Couch-grass and hybrid Couch *Elymus x obtusiuscula* (frequent in Transect 2).

Zone D: dune that varied in width from 15 m (Transect 3) – 43 m (Transect 1). In Transects 1, 4 and 5 there was an eroded cliff (0.4, 1.7 and 0.5 m high respectively) on the seaward side. The grasses Marram *Ammophila arenaria*, Lyme-grass *Leymus arenarius*, hybrid Couch and Rush-leaved Fescue *Festuca rubra* ssp. *arenaria* all featured prominently. Saltmarsh Couch *Elymus atherica*, Sea Bindweed *Calystegia soldanella* (Transect 1), and Shrubby Seablite *Suaeda vera* (Transect 3) were each abundant in one transect. Sea Spurge *Euphorbia paralias* was locally frequent (Transect 5).

Zone E: between the dune and saltmarsh there was a marginal belt 4-12 m wide where Sea Purslane *Atriplex portulacoides* and Purple Glasswort *Salicornia ramosissima* were abundant along with Saltmarsh Couch and Shrubby Seablite. Perennial Glasswort *Sarcocornia parennis* occurred in this zone.

Zone F: the dry saltmarsh in Transect 2 (31m wide) featured Matted Sea-lavender *Limonium bellidifolium*, Rock Sea-lavender *L. binervosum* and Sea Saltmarsh-grass *Puccinellia maritima*. In the wet saltmarsh Common Sea-lavender *Limonium vulgare*, Sea Saltmarsh-grass, Cord-grass *Spartina anglica*, Sea Purslane and Purple Glasswort were prominent.

More notable plants:

Sea Holly *Eryngium maritimum*: present in Transects 5 & 4 (20 and 3 plants) on the landward slope of the dune.

Sea Bindweed *Calystegia soldanella*: abundant on the dune in Transect 1 Zone D.

Sea Rocket *Cakile maritima*: rare on the landward slope of the dune, Transect 5.

Sea Spurge *Euphorbia paralias*: locally frequent on the dune, Transect 5.

Hybrid Cough-grass *Elymus x obtusiuscula*: abundant on the dunes in 5 transects.

Rock Sea-lavender *Limonium binervosum* ssp *anglicium*: frequent in dry saltmarsh depression landward of the dunes, Transect 2.

Matted Sea-lavender *Limonium bellidifolium* (Rare): abundant in dry saltmarsh depression, Transect 2.

Perennial Glasswort *Sarcocornia perennis* (NSc): occasional-rare at the margin of saltmarsh in 4 transects.

One-flowered Glasswort *Salicornia pusilla* (NSc): occasional in dry saltmarsh depression, Transect 2.

b) Titchwell (east sector)

Zone A: a broad sandy lower beach with abundant razor-clam shells, 21 m (west, boardwalk) – 67 m (east, creek) wide. No vegetation.

Zone B: a narrow band of embryonic dune 1-3 m wide. Sparsely vegetated with Sea Couch *Elymus farctus* and Frosted Orache *Atriplex lacineata*.

Zone C: a band of bare sand, 4-14 m wide with little vegetation except occasional Prickly Saltwort *Salsola kali*.

Zone D: a broad fore-dune, 22 m (west) and 45 m (east) wide. Prickly Saltwort (abundant), Frosted Orache (locally frequent), Sea Couch frequent-abundant, Marram *Ammorphila arenaria* (abundant in 2 transects), Sea Sandwort *Honckenya peploides* (frequent in one transect).

Zone E: broad dune including both seaward and landward slopes, 36-45 m wide with a 0.7 m cliff in Transect 2 and 0.5 m high in Transect 3 on seaward side. Well vegetated with Marram (frequent/abundant), Saltmarsh Couch (abundant), Rust-leaved Fescue *Festuca rubra* ssp *arenaria* (locally abundant) and Sea Spurge *Euphorbia paralias* (frequent).

Zone F: saline depression in Transects 1, 3 and 4, 20-48 m wide. Sea Milkwort *Glaux maritima* (abundant), Common Sea-lavender *Limonium vulgare*, (local/frequent). Rock Sea-lavender *L. binervosum* (frequent/abundant in drier areas), Matted Sea-lavender *L. bellidifolium* (rare Transect 3 and 4), Red Fescue *Festuca rubra* (abundant in 2 transects), Long-bracted Sedge *Carex extensa* (Transect 3 – frequent, Transect 4 – scattered).

Zone G: Tidal Lagoon margin in Transects 1 & 2, 10 and 5 m wide. Saltmarsh Couch (abundant), Shrubby Seablite *Suaeda friticosa* (frequent), Rock Sea-lavender (frequent Transect 1), Corn Sawthistle *Sonchus arvensis* (frequent one transect).

Zone H: Inner dune landward of saline depression Transect 3 24 m wide. Sea Spurge (abundant), Saltmarsh Couch (abundant), Red Fescue (frequent), Creeping Bent *Agrostis stolonifera* (locally frequent), Rock Sea-lavender (occasional).

Zone J: Creek Prickly Saltwort (frequent). Shrubby Seablite (abundant seedlings on island), Sea Couch (frequent on island), Frosted Orache (occasional), Purple Glasswort *Solicornia ramasissima* (occasional).

More notable plants:

Yellow Horned-poppy *Glaucium flavum*: 8 non-flowering plants on shingle E of the Creek TM 75514488. Only site recorded.

Sea Rocket *Cakile maritima*: rare in Transects 2 and 3.

Rock Sea-lavender *Limonium binervosum* ssp *anglicum*: saline depression in Transects 1 (scattered), 3 (abundant) and 4 (frequent). Also on margin of Tidal Lagoon and seaward end of track from Centre.

Matted Sea-lavender *Limonium bellidifolium* (Nationally Rare): Transects 3 and 4, saline depression (rare).

Sea Holly *Eryngium maritimum*: 2 plants on dune, Transect 2.

Sea Spurge *Euphorbia paralias*: all 4 transects on dune; abundant on inner dune, Transect 3 (TM 754450); frequent Transect 2 (752450); otherwise occasional.

Long-bracted Sedge *Carex extensa*: in saline depression, frequent Transect 3 (TM 754449) and scattered Transect 4.

b) Titchwell (west sector)

Zone A: bare sand on broad lower beach, 62-80 m wide.

Zone B: embryonic dune, 15-20 m wide. Sparse vegetation with Sea Couch *Elymus farctus*, Frosted Orache *Atriplex lacineata* and Prickly Saltwort *Salsola kali* (Vulnerable).

Zone C: foredune with drift line, 4-15 m. Prickly Saltwort (abundant), Frosted Orache and Sea Couch (both frequent).

Zone D: wide building dune, 22-29 m. Sea Couch and Marram *Ammophila arenaria* (both abundant), Sea Sandwort *Honckenya peploides* (locally abundant), Prickly Saltwort (frequent), Lyme-grass *Leymus arenarius* (frequent in Transect 2). Sea Holly *Eryngium maritimum* (rare in 3 transects).

Zone E: dune ridge 1-7 m wide, with eroded cliff. Cliff 2 m high Transect 3, 3.5 m high Transect 4. Marram (abundant in all 4 transects), Saltmarsh Couch *Elymus atherica* (in 3 transects, abundant Transects 1 & 4), Lyme-grass (frequent Transects 1 & 2), Sea Bindweed *Calystegia soldanella* (frequent Transects 3 & 4), Sand Sedge *Carex arenaria* (Transect 4), Sea Buckthorn *Hippophae rhamnoides* (Transect 4).

Zone F: landward slope of dune. 42-50 m. Not represented in Transect 4 Marram (abundant), Saltmarsh Couch (frequent), Sand Sedge, Shrubby Seablite *Suaeda fruticosa* (frequent saltmarsh margin).

Saline depression between dune and Thornham Point Mound. 4 transects.

Transect A (N-S) 74354510 (dune side). 120 paces across. Rock Sea-lavender *Limonium binervosum ssp anglicum* and Red Fescue *Festuca rubra* abundant on N margin and frequent S margin. Shrubby Seablite *Suaeda vera*, Saltmarsh Couch *Elymus atherica*, Common Sea-lavender *Limonium vulgare* all abundant. Thrift *Armeria maritima* frequent in the wetter centre.

Transect B (N-S) 74204514 (dune) 39 paces across. Rock Sea-lavender on N & S margin, and W limit of depression 74074511, Matted Sea-lavender *Limonium bellidifolia* (Nationally Rare) in centre and S margin, Sea Heath *Frankenia laevis* (frequent) on S margin with Sea Campion *Silene uniflora*, Shrubby Seablite and Red Fescue. Sea Buckthorn *Hippophae rhamnoides* (abundant) and Marram on Thornham Point Mound.

Transect C (N-S) 74134513 west end. 50 paces across. Rock Sea-lavender (abundant on dune margin, occasional S margin), Matted Sea-lavender (abundant in centre and S margin), Sea Heath (abundant S margin), Sea Milkwort *Glaux maritima* (abundant in centre, also S margin).

Transect D (W-E) 74074112-74564500. 783 paces. Matted Sea-lavender (abundant-frequent over 300 paces W end), Rock Sea-lavender (frequent over 100 paces on pebbly sand, W end), Sea Milkwort (frequent-abundant over 225 paces in centre), Common Sea-lavender (frequent-abundant in wetter saltmarsh over 300 paces in centre and east), Thrift (abundant over 260 paces at E end). Other species towards the wetter E end were Greater Sea-spurrey *Spergularia media*, Cord-grass *Spartina anglica* (v local), Sea Plantain *Plantago maritima*, Sea Purslane *Atriplex portulacoides*, Shrubby Seablite (scattered), Purple Glasswort ISalicornia ramosissima and Red Fescue.

Sea Wormwood *Artemesia maritima* occurred with Shrubby Seablite on the drier, sandy E margin 74574502/4.

This large saline depression is an important site for a community associated more with Mediterranean shores and is restricted to North Norfolk in Britain. It does not seem to have been affected detrimentally by the surge and may have benefited from some limited flooding.

Conclusions on Titchwell shore

The Titchwell beach is essentially sandy. The tidal surge on 5/6 December 2013 may have removed up to 30-40 m of dune on the seaward side but it is reforming as embryonic and fore dune. Dune loss most extensive on east sector. The erosion resulted in a sandy cliff forming on the seaward side of the dune, up to 1.7 m high in the Brancaster sector, up to 0.7 m in the eastern sector and up to 3.5 m in the western sector. The board-walk at the seaward end of the path from the Centre to the shore was heaved up and destroyed. The east bank suffered extensive damage and a small amount of leakage occurred at the S end, although the bank was not breached. There was a small amount of over topping along the west bank. Sea water came over the west bank at the Parrinder Hide junction, flooding the path to a depth of c30 cm.

The churning up of the sandy beach and the formation of embryonic and fore dune resulted in a proliferation of Prickly Santwort *Salsola kali* and Frosted Orache *Atriplex lacineata*, Sea Sandwort *Honckeyna peploides* and Sea Couch-grass *Elymus farctus* were also prominent plants in these zones. The dunes were dominated by Marram *Ammophila arenaria*, Saltmarsh Couch *Elymus atherica* and the hybrid Couch *Elymus x obtusiuscula*. Sea Bindweed *Calystegia soldanella*, Sea Spurge *Euphorbia paralias*, Rush-leaved Fescue *Festuca rubra* ssp *oraria/arenaria* and Lyme-grass *Leymus arenaria* were locally frequent. Sea Holly *Eryngium maritimum* was generally scarce on the dunes and in the virtual absence of shingle Yellow Horned-poppy only occurred at a single site.

Botanically the most notable habitat was the fairly dry saline depressions behind the main dune. This supported a nationally very restricted community of plants: Rock Sea-lavender *Limonium binervosum* ssp *anglicum*, Matted Sea-lavender *L bellidiflorum* and Sea Heath *Frankenia laevis* as well as Shrubby Seablite *Suaeda fruticosa*. This community was probably not affected deleteriously by the surge and may have benefited.

Snettisham Pits, West Norfolk (vc 28)

Surveyed 10 June 2014
The shore on the seaward side of the South Pit (1.5 km) plus a short stretch south of the Yacht Club (TF 648322). Five transects surveyed from the low beach to the path at the top of slope.

The 5/6 December 2013 surge flooded the South Pit, destroying the causeway across the South Pit and washing large quantities of shingle from the S end of the South Pit southwards. The Shore Hide was damaged and both The Sanctuary and Roost Hides were destroyed. The boardwalk connecting the southern hides was also destroyed. Part of the roadway allowing vehicle access to the reserve was swept away.

Transect 1 TF 647319 (from bare intertidal mud) at N end of South Pit
2 648317
3 648315 (from accreting saltmarsh just N of former jetty
4 648312 (S of former jetty)
5 647311 (S end of South Pit)

Zone A: Lower beach. Transects 1-4 pebbles, 1-2 m wide. Zone absent in Transect 5.
Accreting saltmarsh 15-20 m wide below pebble zone in Transect 3. Vegetation limited to sparse Glasswort *Salicornia* sp and Annual Seablite *Suaeda maritima* except for accreting saltmarsh where Long-spiked Glasswort *Salicornia dolichostachya* (lower) and Purple Glasswort *S ramosissima* (upper) abundant. Also 10 plants of Perennial Glasswort *Sarcocornia perennis* NSc (one plant in 2013) and Annual Seablite.

Zone B: Mid-beach. Shingle with drift-line.
13-21 m of shingle plus 1-4 m width of drift debris. Zone absent in Transect 5. Vegetation limited to drift-line in Transects 3 & 4. Sea Mayweed *Tripleurospermum maritimum* (3 of 4 transects), Sea Sandwort *Honckenya peploides* (2 transects), Yellow Horned-poppy *Glaucium flavum* (2 flowering plants, many rosettes Transect 1), Sticky Groundsel *Senecio viscosus* (frequent Transect 3), Spear-leaved Orache *Atriplex prostrata* (abundant drift line Transect 4), Prickly Saltwort *Salsola kali* (Vulnerable) and Shore Orache *Atriplex littoralis* (drift-line, Transect 3).

Zone C: Upper beach. Well vegetated shingle.
4-19 m wide. Transects 1-4, Sea Sandwort (abundant 3 transects), Sea Spurge *Euphorbia paralias* (4 transects, abundant Transect 1, sparse Transect 3 but extending over 26 m along beach), Sea Mayweed (3 transects, frequent/abundant in 2), Yellow Horned-poppy and Bitter Stonecrop *Sedum acre* (both frequent Transects 2 & 4), White Stonecrop *Sedum album* and Sea Campion *Silene uniflora* (both in Transects 1 & 2), Saltmarsh Couch *Elymus atherica* (frequent but small, Transect 4), Curled Dock *Rumex crispus* (occasional 4 transects).

Zone D: Seaward slope. Vegetated shingle.
12 m width Transect 1 (N end) increasing to 28 m Transect 4. Sea Mayweed (abundant in 2 transects), Marram *Ammophila arenaria* (abundant Transect 3, frequent Transect 4), Saltmarsh Couch (abundant Transect 4), Curled Dock (2 transects, abundant Transect 2), Common Rest-harrow *Ononis repens* (abundant Transect 3), Yellow Horned-poppy (rosettes frequent Transect 1).

Zone E: Top of slope adjoining the path.
On Transect 2, 10 m wide with drift-line. A 4 m wide dry shelf on Transect 4. In Transects 1 and 3 this zone incorporated with Zone D. White Stonecrop (abundant Transect 2, frequent Transect 4), Bramble *Rubus fruticosus* agg and Rose *Rosa* sp (each in 3 transects), Teasel *Dipsacus fullosum* (rosettes) and Rose-bay Willowherb *Chamerion angustifolium* (Transects 1 & 2). On dry shelf in Transect 4, Viper's Bugloss *Echium vulgare*, Carlina Thistle *Carlina vulgaris*, Fern-grass *Catapodium rigidum* and Bearded Fescue *Vulpia ciliata*.

Transect 5 at the S end of South Pit
Saltmarsh (14 m wide) with glasswort *Salicornia* sp (abundant), Sea Spurry *Spergularia media* (frequent), Shrubby Seablite *Suaeda vera* (frequent), Bare shingle (9 m) with little vegetation. Rather large open shingle (17 m) with Sticky Grounsel *Senecio viscosus* (abundant), Sea Campion (frequent), White Goosefoot *Chenopodium album* (frequent), Yellow Horned-poppy and several ruderal plants such as Common Fumitory *Fumaria officinalis* and Black-bindweed *Fallopia convolvulus*.

More notable plants:
Perennial Glasswort *Sarcocornia perennis* (NSc): an accreting saltmarsh N of former jetty. TF 64803154. 2013, one plant; 2014, 10 plants.

Shrubby Seablite *Suaeda vera* (NSc): 2002 & 2014, a large patch on bank at S end of South Pit. Scattered small plants in accreting saltmarsh N of former jetty. (2014). A large patch S of the Yacht Club.

Sea Spurge *Euphorbia paralias*: 2006, a few small plants on shingle beach at N end of the reserve, TF 6474399. By 2013 there were 2 extensive patches (a) extending over 82 paces, 6473193; (b) over 116 paces 64803184. In 2014, patch (a) was 73 x 18 paces in extent, 647753195 (N) – 64793189 (S); (b) was 105 paces long, 64793167 (N) – 64813161 (S). There were 3 plants on the upper beach in both 2013 and 2014.

Bearded Fescue *Vulpia ciliata* (NSc): 2014, on dry shelf at the top of the seaward slope near path TF 64803168, with Fern-grass and Sand Cat's-tail *Phleum arenarium*,

Prickly Saltwort *Salsola kali* (Vulnerable).
2009: abundant along much of the shore, close to the tide-line. TF 64783190-64793172; sparser further S to 64793163. Also 64813159-64763120, 63603280 and near Yacht Club 646334.

2013: small quantity on strandline, 64773113.

(2014): not recorded in any of the 5 transects that covered the beach.

Appears to have declined, but lack of sandy substrate perhaps the cause. No evidence that the surge had a noticeable effect.

Sea Kale *Crambe maritima*
1996: 9 plants on shingle N & S of the Yacht Club, TF 646334-647326.

(2002): None seen at the above site.

2014: 18 plants well up the shingle beach, S of the Yacht Club, 64753270. 1 large plant Snettisham Scalp, 64653272. Stiring up the shingle by the surge may have been beneficial to this species.

Frosted Orache *Atriplex lacineata*
2002: scattered along strandline on shingle beach, including S of the Yacht Club.

2014: a total of 7 plants at 6 sites on lower beach, including in Transects 3, 4 and 5.

Sea Rocket *Cakile maritima*
2014: 2 single plants on shingle at south end, including Transect 5.

Yellow Horned-poppy *Glaucium flavum*
2002: along strandline N of the N end of South Pit northwards.

2013: abundant along strandline on beach S from the gate at N end of reserve.

2014: frequent in Transects 1 (Zones B and D), Transects 2 and 4 (Zone C) including many non-flowering rosettes.

Sea Bindweed *Calystegia soldanella*
2013 & 2014: locally flowering in quantity on SE-facing bank S of gate at N end of the reserve, TF 64813193.

Hoary Cinquefoil *Potentilla argentea*
2013: 20 plants on W-facing bank near slabs of concrete on W side of South Pit, TF 64843169. New species for the reserve.

2014: 31 plants at 7 sub-sites on both E and W side of track in approximately the area of the 2013 site, 64843157-64843170. Probably not affected by the surge.

Glabrous Cat's-ear *Hypochaeris glabra* (Vulnerable).
2013: 4 plants on W facing bank near slabs of concrete on W side of South Pit TF 64843169.

2014: 3 plants at the same site.

Knotted Pearlwort *Sagina nodosa*
2002: a 7 x 7 patch in damp area on E side of the South Pit just S of the causeway, TF 64963133. Not recorded subsequently. A scarce plant on the Norfolk coast.

Upright Chickweed *Moenchia erecta*
1996: east side of South Pit, south of causeway, TF 650309 (Stuart Fisher). Not recorded subsequently.

Red Hemp-nettle *Galeopsis angustifolia* (NSc *arachaeophyte*, Critically Endangered): 3 main sites on shingle back from the beach.
a) East side of South Pit, both north & south of the causeway, TF 64943140-64953139.
b) Within fenced area south of South Pit, 64803069 (former shingle extraction pit) and 64813069 (close to a small elder bush).
c) Unfenced area south of the Sanctuary Hide, south of the South Pit 64933044.

Sites (b) and (c) monitored most years since 1996; (a) since 2009.

Number of plants				
	Site			
	a	**b**	**c**	**Total**
2009	1300	1444	262	2006
2010	1045	710	763	2518
2011	Several thousand	521	114	3660 +
2012	Thousands small plants	168	1078	5000 +
2013	2700	204	30	2934
2014	771	433	273	1477

Heavily grazed by rabbits or hares in some years as in 2003 and 2009, resulting in a major reduction of plants later in the season and flowering on secondary growth.

2003	b&c	993 plants (24.7)	603 (14.8)	116 (13.9
2009	a, b & c	3006 plants (28.7)	452 (8.8	

In 2006 there were no plants at site (c) where the shingle had become overgrown with False-oat *Arrhenatherum elatius* , Sea Campion *Silene uniflora* and Red Fescue *Festuca rubra*. In the autumn of that year a plot 2 x 2 m had vegetation removed and was scarified.

	2007	**2009**
Scarified	22	177 plants
Control	Nil	7 plants

There is no conclusive evidence that the populations at the 3 sites were detrimentally affected by the surge in December 2013, though the shingle at both sites (b) and (c) had been considerably churned up.

Hoary Mullein *Verbascum pulverulentum* (NSc).
Up to 2002/03 its distribution was largely around the chalets to the north of the reserve, particularly in 2 gardens overlooking the North Pit (TF 64743317).

1996: 2306 plants in all stages, mostly around the chalets close to Cockle Road (647332-648327).

1997: Compt 2 – 17 plants (5 flowering); Compt 3, 2 plants (1 flowering), Compt 4, 184 plants (19 flowering). Total 203 plants (23 flowering).

2003: gardens of 2 chalets (64743317), 258 plants (86 flowering). South of gate onto the reserve (647322-647324), 10 plants (8 flowering).

2009: shingle ridge, south of gate just inside the reserve (64793193), 11 dead inflorescences from 2008, hundreds of rosettes.

2010: south of gate within the reserve (same as 2009 site) 410 plants (159 flowering); by hide at south west of South Pit (64773089), 3 flowering plants.

2014: south of gate, 6463195, only 19 plants (3 flowering); near concrete blocks at north end of South Pit (64823124), 1 rosette; south west end of South Pit (64823124), 1 flowering plant. The December 2013 surge seems to have checked the spread on the reserve.

Bird's-foot Clover *Trifolium ornithopodiodes*
2014: one small plant in trampled path by corner at south end of South Pit, TF 64733094.

Prickly Sedge *Carex muricata*
2014: a single clump on dry shingle at south end, TF 64903049.

Japanese Rose *Rosa rugosa* (naturalised)
2014: a large patch 14x7 m near path at the top of the shore opposite South Pit, TM 64803168).

Rock Sea-lavender *Limonium binervosum* ssp *anglicum*
1998: c8,000 plants (80% non-flowering young) on shingle back from the beach south of the Yacht Club TF 648322-648325.

2009: locally abundant extending over 103 m, south of the Yacht Club, The Scalp, 64653272-64703266.

2013: thousands of plants over 100x10 m strip on dry sandy shingle to east of a large patch of Shrubby Seablite to south of Yacht Club.

2014: scattered over a strip 104 m long at same site as above 64663271 (north) – 64693266 (south). Some decrease in number of plants but not distribution since the December 2013 surge.

Matted Sea-lavender *Limonium bellidifolium* (Rare)
1998: 7 plants over 15 m (diam) in depression among Shrubby Seablite south of Yacht Club, TL 647327.

2000: 10 plants at the above site.

2009: 3 plants at the same site.

(2010): no plants at the site.

2013: 4 plants on dry sandy shingle to north of an extensive area of Shrubby Seablite, south of the Yacht Club 64643272. Vulnerable to drying at the depression and being outcompeted by the *Suaeda*.

(2014): no plants at the above site.

This species has been lost from other sites around the Wash.

Effects of the surge on the flora

Benefits:
Reappearance of Sea Kale in the vicinity of the Yacht Club.

Declines:
Prickly Saltwort appears to have (temporarily) disappeared, but was already decreasing, possibly as the result of sandy substrate on the beach being replaced by shingle.

Red Hemp-nettle suffered a decline in the number of plants at site a by the causeway across South Pit, but the populations at the other two sites at the south end increased from the 2013 counts.

The spread of Hoary Mullein onto the north end of the reserve seems to have been checked.

Rock Sea-lavender that was locally so abundant south of the Yacht Club was only scattered at the site in 2014.

Little change:
Sea Spunge – the two possible patches on the beach remained from 2013 to 2014.

Sea Bindweed – a patch south of the date at north end of the reserve continues to flourish.

Hoary Cinquefoil – discovered on the reserve in 2013 showed a spread in 2014, but this was unlikely to have resulted from the surge.

Can intensive agricultural landscapes favour some raptor species? The Marsh harrier in north-eastern Spain

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Abstract

Agricultural intensification over the last 60 decades has been linked to decreases in biodiversity and in the breeding populations of several avian species in farmlands. However, agricultural intensification has not affected all species in the same way and transformed landscapes can still provide suitable habitats for species tolerant to some degree of anthropogenic change. Understanding habitat selection in man-made landscapes is a pre-requisite to effective management and conservation of the species that use them. However, habitat-related choices made by individuals occupying these landscapes are often difficult to explain, as the cues they use may be decoupled from the ecological context in which they evolved. Here, we investigated nesting habitat selection in a ground-nesting raptor breeding mainly in wetlands, which, unlike many other species occupying farmlands, has experienced a population increase in some agricultural regions. We used multivariate analysis to assess the extent to which habitat characteristics, human disturbance or proximity to other occupied sites influenced nesting-site occupancy. Our results indicate that Marsh harriers occupied breeding sites according to habitat cues obtained at two complementary spatial scales (i.e. the breeding site and the foraging area). This study indicates that a raptor species that uses human-made structures such as ponds for breeding while exploiting their surrounding crops for hunting can take advantage of agricultural intensification. Environmental disturbances are often viewed only in terms of their harmful impacts on the affected species. However, human activities are causing many types of alterations in natural landscapes that can be exploited by certain species with positive responses towards these transformations. The adaptation of some raptors to human-altered environments, such as the Marsh harrier in our study area, raises doubts regarding the appropriateness of using such species as indicators of natural habitat quality.

Introduction

Agricultural intensification over the last six decades has been linked to decreases in biodiversity and in breeding populations of several avian species in farmlands (Tucker & Heath, 1995; Chamberlain *et al*., 2000; Donald, Green & Heath, 2001). Contributing factors to the overall impact of intensification include pesticide use, removal of hedges and other uncultivated areas to produce larger fields, earlier harvesting dates and an increase in monocultures (Newton, 2004). Generally speaking, such agricultural practices have resulted in losses of habitat quality and heterogeneity that have negatively affected avian species directly through increments in mortality or reproductive failures or indirectly by reducing or degrading their hunting and nesting habitats (Benton, Vickery & Wilson, 2003; Newton, 2004). In this context, studies of habitat selection in farmlands have been central in conservation biology and have been described as an essential tool to assess the habitat requirements of a species and to develop effective management strategies (e.g. Blanco, Tella & Torre, 1998; Tella *et al*., 1998; Arroyo, García & Bretagnolle, 2002; O'Connell & Yallop, 2002; Carrete & Donázar, 2005; Arroyo *et al*., 2009).

Agricultural intensification has not affected all species in the same way, and transformed landscapes still provide suitable habitats for species tolerant to anthropogenic alterations. For example, in intensively managed lands, where artificial ponds and irrigated crops have replaced natural wetlands or created new ones, some bird species have benefitted through the creation of new foraging or breeding habitat (Czech & Parsons, 2002; Tourenq *et al*., 2004; Sánchez-Zapata *et al*., 2005; Moreno-Mateos, Pedrocchi & Comín, 2009; Sebastián-González, Sánchez-Zapata & Botella, 2010). The Marsh harrier *Circus aeruginosus* is a medium-size raptor that breeds mainly in wetlands and behaves as an open-habitat hunter (Cramp &

Simmons, 1994). The European Marsh harrier population underwent a dramatic population decline from 1960 to 1980 due to drainage of wetlands, contamination and direct persecution, but recovered in recent decades, when the species underwent a moderate increase, $<10\%$, (Cramp & Simmons, 1994; BirdLife International, 2004). Contrary to other species occupying farmlands, the recovery of this species has been very noticeable in some agricultural regions, even in areas dominated by intensified crops. Within some of these regions such as the Iberian Peninsula, the increase in the Marsh harrier population has been accompanied by an increase in its geographic range, with some pairs occupying artificial ponds and reservoirs related to agricultural practices. However, the extent to which these recent occupations are due to habitat characteristics, to the proximity of other occupied sites or to a combination of these factors, is unclear. Understanding the attractiveness and quality of wetlands for wildlife in human-transformed environments is vital to the success of attempts to reconcile development and biodiversity conservation (Froneman *et al*., 2001; Moreno-Mateos *et al*., 2009; Sebastián-González *et al*., 2010). Furthermore, the expanding Marsh harrier population provides an opportunity to identify some of the anthropogenic factors involved in apparently natural invasion processes that are currently favouring worldwide biotic homogenization (McKinney & Lockwood, 1999; Rice & Pfenning, 2008).

Here, we investigate the factors influencing nesting-site occupancy in an expanding population of Marsh harriers located in an agricultural area in the Iberian Peninsula, the Catalan Ebro Basin (Fig. 1a), where the species has increased from eight breeding pairs in 1997 to 53 in 2008 (Fig. 2). For this purpose, we establish the relative importance of habitat characteristics and social behaviour in the occupancy rate of breeding sites at two spatial scales. The results are discussed within the current scenario of agriculture intensification and its potential effects on biodiversity.

Materialsandmethods

Studyareaandfieldprocedures

The study area comprises 7300 km^2 within the Catalan Ebro Basin, north-eastern Spain (Fig. 1a and b). The landscape is mostly low-lying and flat, broken by discrete ranges of small hills (0–400 m a.s.l.), and has a semiarid Mediterranean climate. The annual precipitation is 300–500 mm, with most rain occurring in spring and autumn, and an annual thermal amplitude of 19–20 °C (Bosch, 2004). Traditional agriculture in this area comprises extensive cultivation of cereal crops separated by field margins, where harvested fields remain uncultivated for 1 or more years (fallow system) and are grazed extensively by livestock (Suárez, Naveso & de Juana, 1997). During the 20th century, agriculture in this area underwent several dramatic changes, including the replacement of traditionally cultivated cereals (e.g. spring-sown wheat and barley) with a variety of alternative crops (e.g. winter-sown wheat and barley, alfalfa, maize, sunflowers, deciduous fruit trees, horticultural crops and rice), the establishment of irrigation systems and the disappearance of the fallow system and field margins (Tella *et al*., 1998). Now, the study area is an agricultural mosaic composed of arable, non-irrigated cereal crops (mainly winter-sown wheat and barley), irrigated fields (mainly alfalfa, winter-sown wheat, winter-sown barley and some sweet corn), dry fruit trees (mainly olive and almond trees) and irrigated fruit trees (mainly peach, pear, apple and nectarine trees). A total of 617 wetlands have been described over the study area, of which *c*. 92% are artificial ponds and reservoirs related to agricultural irrigation (A. Varea, unpubl. data).

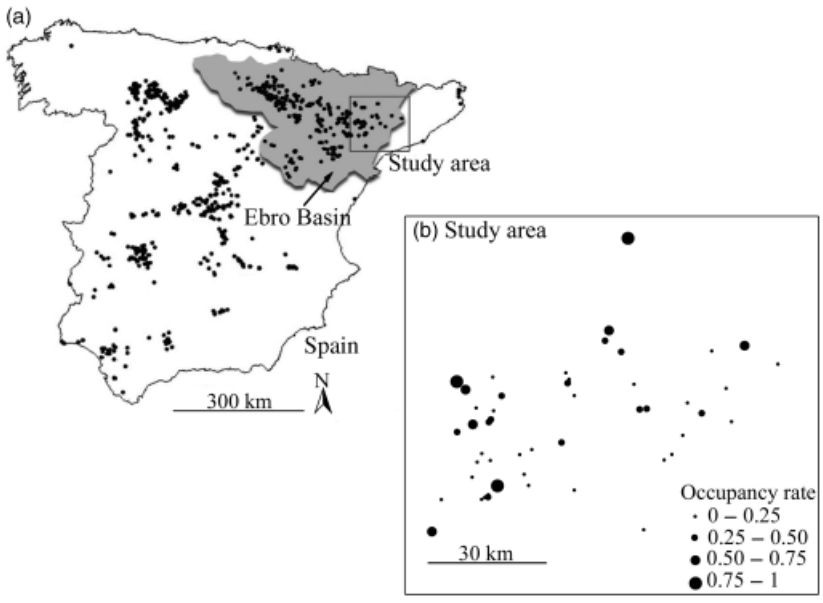


Figure1 (a) Distribution of the breeding population of the Marsh harrier *Circusaeruginosus* in Spain in 2006 (Molina & Martínez, 2008). The Ebro Basin is shown in grey; (b) occupancy rate of monitored breeding sites in the study area (Catalan Ebro Basin, Spain, 1997–2008).

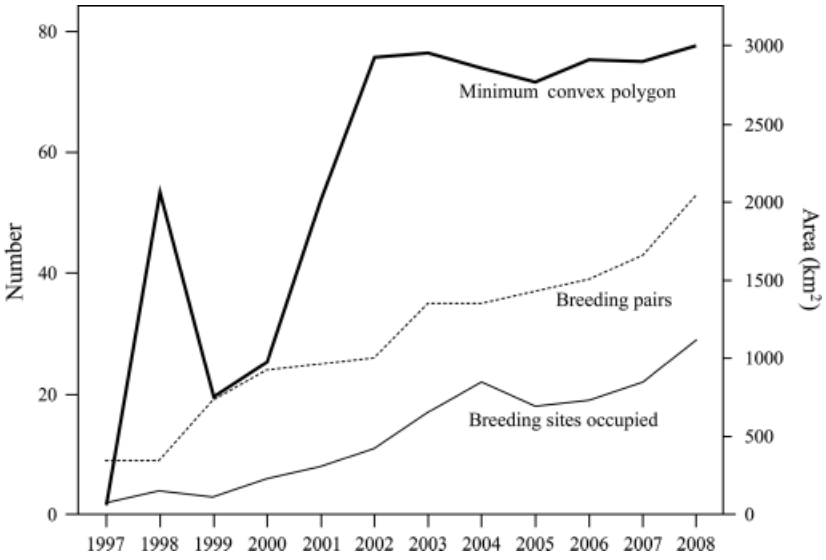


Figure 2 Population trend for the breeding population of Marsh harrier*Circusaeruginosus* in the Catalan Ebro Basin, Spain, showing the number of breeding pairs, the number of breeding sites occupied and the minimum convex polygon area including the occupied breeding sites recorded each year (Source: Departament de Medi Ambient i Habitatge de la Generalitat de Catalunya and own data).

The breeding population of Marsh harriers in the study area was systematically monitored from 1997 to 2008 (Fig. 2). Each year, known breeding sites and other areas potentially suitable for breeding were visited repeatedly to determine occupancy, looking for signs of breeding behaviour (i.e. territorial displays or deliveries of nest material or prey items to the nest). Potentially suitable breeding sites were considered to include all wetlands (defined as artificial ponds, reservoirs, marshes and parts of rivers or watercourses; see Table 1) with the presence of tall standing reeds *Phragmites australis*, reedmace *Typha angustifolia* or other dense emergent aquatic vegetation (Cramp & Simmons, 1994). Visits began in early April, when nest building started, and continued until fledging (May–July). A breeding site was considered to be occupied when at least one breeding pair laid eggs.

Characteristicsofthebreedingareas

The location of monitored breeding sites and associated habitat variables were entered into ArcView 3.2 Geographic Information System. We measured a total of 14 habitat variables within each monitored wetland and within a 3 km radius around the wetland to describe nest site and landscape characteristics, respectively (Table 1). Three kilometres is the average home-range size of breeding Marsh harriers estimated on the basis of radio-tracking data (Cardador *et al.*, 2009).

Variables measured at the wetlands were obtained from 1:5000 aerial photographs from 2005 (Institut Cartogràfic de Catalunya) ground-truthed in the field during 2006 (Table 1). The type of wetland was assessed in the field, while land uses and variables describing human disturbance such as distance from wetlands to the nearest villages, roads and paths were obtained from 1:50 000 digital land-use maps (Departament de Medi Ambient i Habitatge de la Generalitat de Catalunya; Table 1).

Proximity to other occupied sites was estimated as the linear distance from wetlands to the nearest occupied site (NND), and through an isolation index (S) describing the relative position of a given wetland within the spatial distribution of the breeding population (Carrete *et al.*, 2006). Because our Marsh harrier breeding population was the continuation of a larger breeding population located in the west of the Ebro basin (Fig. 1a), for calculations of NND and S values, we took into account Marsh harrier breeding locations both within and outside of our study area. The NND of each wetland could be calculated annually, and we used their average value as the input data. The S was taken from metapopulation theory relating to connectivity, and calculated for wetland i as $S_i = \Sigma\exp(-d_{ij}) \times A$j (with $i \neq j$), where d_{ij} is the linear distance between wetland i and j, and Aj is the number of breeding pairs in the wetland j, j representing all known occupied wetlands. Because this index is formulated to describe connectivity (Moilanen & Hanski, 1998), smaller values indicate increasing isolation. Because accurate information of the whole Marsh harrier breeding population outside the study area was only available for 2006 (data from Dirección General de Desarrollo Sostenible y Biodiversidad, Zaragoza, and SEO/BirdLife) and because the global geographic distribution of the population (and thus S) has remained unchanged during recent years (based on the comparison between distribution maps of the Marsh harrier breeding population from 1998 and 2006; Martí & Moral, 2003; Molina & Martínez, 2008), we used S from 2006 as a single estimate of isolation. This approach is justified by the fact that, for the territories in this study, estimates of S in 2006 were correlated with estimates of this parameter in all other years (Spearman's rank correlation range = 0.263–0.944; P-value range: <0.0001–0.031).

Analysis

We used logistic regressions (using the genmod procedure of SAS 8.0; SAS Institute, 2004), with a stepwise procedure, to analyse the factors affecting occupancy of wetlands by

Table 1 Variables used in models to analyse the occupancy of Marsh harriers *Circus aeruginosus* in Catalonian Ebro Basin, Spain, 1997–2008

Variables	Definition
Habitat variables	
Breeding-site level	
Type	Artificial pond (included constructions of small dimensions <13 ha to store water related to agricultural practices), reservoir (included water bodies originated by a floodgate), river (included junctions of rivers and meanders of permanent rivers), watercourse (included reedbeds in natural channels through which water flows only occasionally, during periods of maximum rain), inland natural wetland (included inland marshy terrains and salt marshes where water availability drastically decreases or even dries completely during spring and summer).
Vegetation	Area of emergent vegetation at the wetland (ha)
Shape	Length of emergent vegetation edge per surface of emergent vegetation ($m\,ha^{-1}$)
Water	Area of open water (ha)
Landscapelevel	
Intensified-herbaceous	Extent of intensified, irrigated herbaceous crops within 3 km of the wetland (%)
Intensified-tree	Extent of intensified, irrigated fruit trees within 3 km of the wetland (%)
Traditional-herbaceous	Extent of traditional, non-irrigated herbaceous crops within 3 km of the wetland (%)
Traditional-tree	Extent of traditional, non-irrigated fruit tree crops within 3 km of the wetland (%)
Vineyard	Total extent of intensified vineyards within 3 km of the wetland (%)
Other	Total extent of unsuitable habitats for the species (i.e. urban areas and forest) within 3 km of the wetland (%)
Village	Distance to the nearest village (m)
Road	Distance to the nearest road (m)
Path	Distance to the nearest path (m)
Proximity to other occupied sites	
NND	Distance to the nearest occupied wetland (km)
S	Isolation of the wetland with respect to the other occupied ones

NND, nearest neighbour distance.

Marsh harriers (binomial error distribution; logit-link function). As wetland sites comprise only emergent vegetation and open water, the inclusion of these two variables means that the size of the entire site was accounted for in the models (Table 1). We used the occupancy rate (number of years occupied/total years monitored) as the dependent variable. For each significant model ($P<0.05$), we calculated the percentage of deviance explained ($100\times(\text{deviance}_{\text{null model}}-\text{deviance}_{\text{model}})/\text{deviance}_{\text{null model}}$). As neighbouring wetlands could have similar environmental conditions due to their spatial autocorrelation, we added, in every model, a spatial term composed of the X and Y coordinates (UTM Projection), and all of their interactions terms up to the third order (Legendre & Legendre, 1998). However, to avoid artificial increases in the explained variation, we first tested the nine explanatory terms in our logistic regression models using a stepwise procedure and we only retained the significant ones (Borcard, Legendre & Drapeau, 1992).

Following Carrete *et al*. (2007), we used deviance partitioning to take account of co-linearity between spatially related variables. Deviance partitioning entails the calculation of incremental improvement in models due to the inclusion of a variable in every possible model incorporating that variable. This allows the 'pure' effect of each explanatory variable to be separated from joint effects that cannot unambiguously be attributed to one variable or another (Fig. 3a). For a more detailed description of this method, see Carrete *et al*. (2007).

Finally, we tested whether the relationship between area of emergent vegetation (vegetation) and occupancy rate could be accounted for by a null model involving only random settlement with respect to vegetation area. Following Gillings & Fuller (2001), we used a randomization procedure that produced the expected occupancy rate for locations of different sizes (for each location, this procedure was repeated 100 times). Thus, our database included a set of observed values of occupancy rates (one for each locality) and a set of expected values derived from random settlement (100 for each locality). A generalized linear model (GLM) (binomial error distribution; logit-link function) with occupancy rate as the dependent variable was used to determine whether there was a significant difference in the relationship between vegetation area and occupancy rate for the observed set and the random set of data. Specifically, we tested the potential interaction between 'vegetation' and 'type of data' (a categorical variable with two levels: observed/random).

Results

Wetlandscharacteristicsandoccupation patternofthebreedingpopulation

In total, 51 wetlands within the study area had the minimum requirements described for Marsh harrier (i.e. the presence of dense emergent aquatic vegetation) and were monitored during the study period (Fig. 1b). Wetland sizes ranged from 0.5 to 287 ha, but 88% of them were smaller than 15 ha, with an average size of 4.9 ± 4.1 ha. Only 32% of monitored wetlands were of a natural origin, while 39% were artificial ponds and 29% were reservoirs related to intensive

agriculture (for definitions of artificial ponds and reservoirs, see Table 1). Emergent vegetation was mainly composed of a mixture of reed and reedmace, with reed being the dominant species in *c*. 90% of wetlands. Thirty-five out of the 51 monitored wetlands were occupied at least once from 1997 to 2008 (Fig. 3, Table 2). During this period, we recorded 375 breeding attempts, 354 of which resulted in egg laying. For wetlands occupied at least once during the study period, the mean number of breeding pairs per year of occupation was 1.45 ± 1.79 (range 1–11.50, $n = 35$).

Occupancy models for the breeding population

Our results show that Marsh harriers occupied breeding sites using habitat cues obtained at two complementary spatial scales. Proximity to other occupied sites seemed not to strongly affect occupancy rates (NND: $\chi^2 = 0.04$, $P = 0.84$; *S*: $\chi^2 = 0.09$, $P = 0.76$).

The habitat model explained 67% of the deviance in the dataset, a value that increased to 71% when the latitude (*Y*) was also included. The deviance partitioning analysis shows that the area of emergent aquatic vegetation had the greatest effect on the occupancy rate of wetlands by Marsh harriers, its pure effect accounting for *c*. 65% of the total deviance explained (Fig. 3b). A GLM showed that the observed relationship was significantly greater than a null model would predict (interaction vegetation × type of data: $\chi^2 = 6.20$, $P = 0.013$), indicating that the observed increase in the occupancy rate with vegetation area was steeper than expected and could not arise solely by a random settlement pattern (Fig. 4). The pure effects of the type of wetland (25%) and the percentage of intensified herbaceous crops in the surroundings (17%) were also important in explaining variability in occupancy (Fig. 3b). A smaller amount of deviance (< 5%) was explained by each of the joint effects of these variables (d, e, f and g). Thus, the occupancy rate of wetlands by Marsh harriers increased with their area of emergent aquatic vegetation and the percentage of irrigated herbaceous crops in their surroundings, with a minimum at inland-natural wetlands (see Table 3).

Analysis of changes in mean productivity of locations with respect to the habitat characteristics measured in the study suggested that no significant relationships existed between this demographic parameter (i.e. mean number of young fledged per breeding pair) and habitat attributes

Table2 Characteristics of breeding sites occupied at least once during the study period, Catalan Ebro Basin, Spain, 1997–2008

Variable	Mean ± SD	Range	Units
Habitat variables			
Breeding-site level			
Vegetation	8.4 ± 19.6	0.5–103.2	ha
Shape	551 ± 290	105–1268	m ha^{-1}
Water	18.5 ± 49.8	0–224	ha
Landscape level			
Intensified-herbaceous	41.4 ± 31.5	0.0–99.5	%
Intensified-tree	18.3 ± 22.4	0.0–83.4	%
Traditional-herbaceous	20.0 ± 27.4	0.4–94.8	%
Traditional-tree	3.0 ± 5.4	0.0–16.9	%
Vineyard	4.8 ± 11.4	0.0–41.4	%
Other	12.6 ± 17.3	0.0–74.8	%
Human pressure			
Village	2348 ± 1107	75–4235	m
Road	66 ± 129	0–631	m
Path	40 ± 79	0–280	m
Proximity to other occupied sites			
Distance	13.8 ± 8.6	3.8–32.2	km
Isolation	0.13 ± 0.26	0–1	

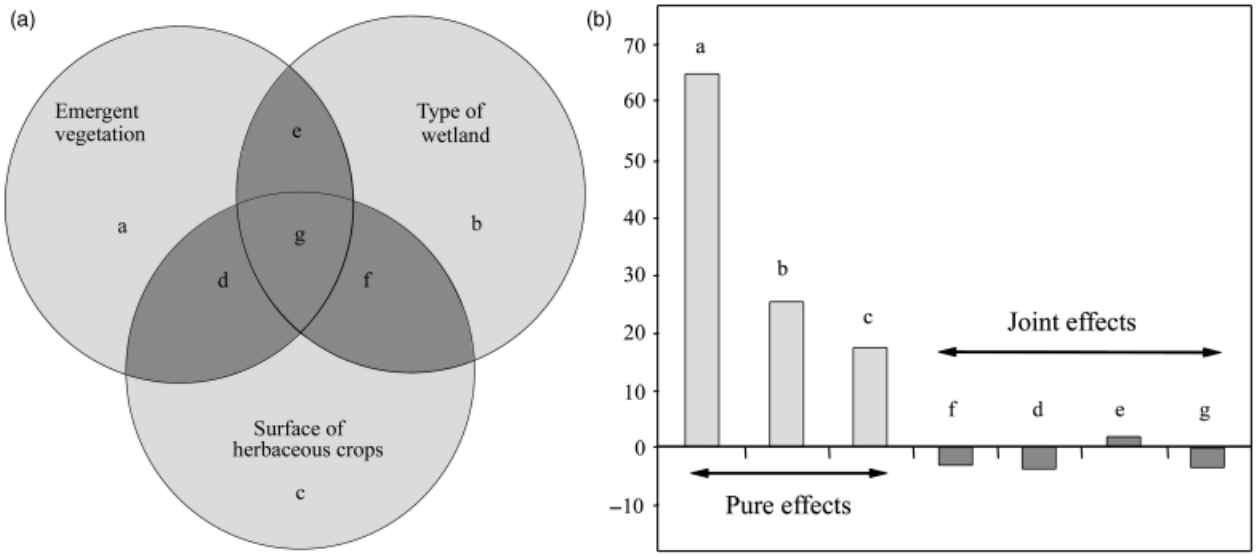


Figure3 (a) Deviance partitioning analysis for the occupancy rate of breeding sites by Marsh harriers *Circus aeruginosus*. (b) Percentage of variability in occupancy rate explained by the pure and joint effects of habitat variables.

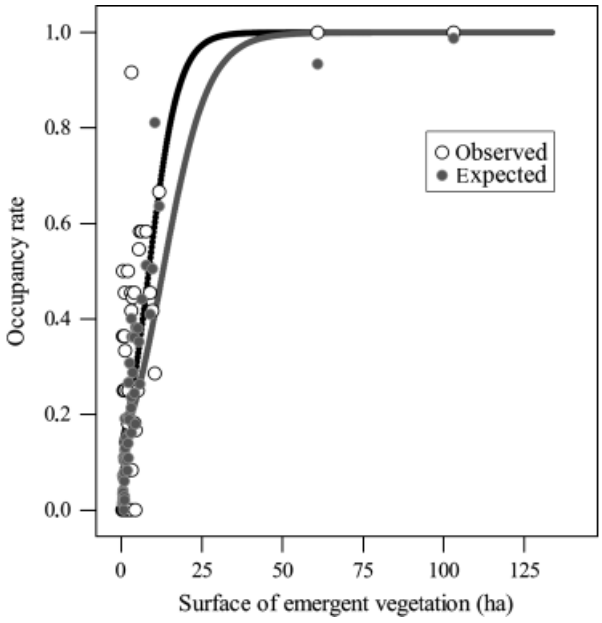


Figure4 Effect of surface of emergent vegetation on the occupancy rate, derived from fitted values from GLM estimates (binomial error distribution, logit-link function). The observed trend is shown with a black line. The grey line indicates the expected trend under a null model in which occupancy rate variation arises solely by individuals settling randomly with respect to vegetation area. Observed occupancy rates and mean expected occupancy rates from 100 replicates (see ''Materials and methods'') are also shown.

(GLMs, normal error distribution, identity link function: all $P > 0.45$).

Discussion

Factors affecting occupancy rate: can human-made landscapes provide suitable breeding habitat for Marsh harriers?

When choosing a site at which to settle, individuals need to simultaneously consider the quality of available sites in terms of their environmental conditions, resource availability and the presence of predators and/or parasites (Martin, 1993; Penteriani, Gallardo & Roche, 2002). However, this quality can be modulated by the presence of conspecifics, which may either decrease or enhance the appeal of a site (Stephens & Sutherland, 1999). In our study model, the occupancy rate of wetlands by Marsh harriers was mainly explained by the independent effect of habitat features acting at two spatial scales, that is, the breeding site (area of emergent aquatic vegetation and type of wetland) and the foraging area (percentage of intensified, irrigated herbaceous crops in the surroundings). Neither human pressure variables nor proximity to other occupied sites seemed to be important determinants of occupancy rates in this expanding population.

Table3 Generalized linear model obtained for the occupancy of breeding sites by Marsh harriers *Circus aeruginosus* in the Catalonian Ebro Basin, Spain (1997–2008)

Parameter	Estimate	SE	χ^2	P
Intercept	−148.06	48.85	9.19	0.0024
Y	0.00003	0.00001	8.89	0.0029
Type (inland-natural wetland)	−3.69	0.92	16.18	<0.0001
Type (artificial pond)	−0.06	0.37	0.03	0.8743
Type (reservoir)	−0.47	0.37	1.59	0.2068
Type (river)	−1.07	0.73	2.13	0.1443
Type (watercourse)	–	–	–	–
Vegetation	0.28	0.05	27.3	<0.0001
Intensified-herbaceous	0.02	0.01	22.01	<0.0001

d.f. = 41.

Nest protection and food availability are key determinants of habitat selection patterns in different avian species (Martin, 1993; Penteriani *et al*., 2002; Sergio, Pedrini & Marchesi, 2003). In our study species, both factors seem to influence occupancy. Indeed, occupancy rate increased with emergent vegetation area steeper than expected by a random settlement pattern. This relationship was particularly apparent in wetlands with small and intermediate surfaces of emergent vegetation (up to *c*. 50 ha). The occupancy rate was higher in wetlands with larger surfaces of emergent vegetation, perhaps because harrier breeding attempts in such sites experience lower rates of failure due to terrestrial predators, disturbance by humans or local flooding events (Bavoux *et al*., 1989; Fernández & Azcona, 1993; Buczek & Keller, 1994; Stanevicius, 2004; Eglington *et al*., 2009). At a landscape scale, harriers responded positively to the area of irrigated herbaceous crops, suggesting that they offer a larger abundance or accessibility to prey (Preston, 1990). This interpretation is reinforced by a previous finding showing how radio-tracked Marsh harriers occupying intensified habitats tended to have smaller home ranges than birds nesting within non-irrigated ones (Cardador *et al*., 2009). Intensified farmlands in our study area included cereals (basically irrigated wheat and barley), alfalfa and some maize crops. Some of these crops, such as intensified cereals and maize, are used infrequently by Marsh harriers, probably because the height and uniformly dense growth that characterize these crops for much of the summer result in low availability of prey to hunting harriers (Cardador & Mañosa, in press). However, Marsh harriers may be taking advantage of regularly mowed fields (intensively managed alfalfa), which offer an adequate vegetation structure for hunting during a major part of the year (Ursúa, Serrano & Tella, 2005; Kitowski, 2007). Furthermore, intensified herbaceous crops in our study area, mainly alfalfa fields, seem to hold higher densities of small mammal prey, especially Mediterranean Pine Vole, than do more extensive crops (A. Varea and A. Folch, unpubl. data). More specific studies on hunting habitat selection are necessary to determine whether and to what extent managed herbaceous crops are preferred hunting habitats of Marsh harriers.

The occupancy rate at natural wetland sites was lower than for any other habitat type, and may be a consequence of their drainage pattern. In the study area, water reserves at natural wetlands decrease drastically, or even dry out completely, during spring and summer, when harriers breed. Thus, their emergent vegetation is less developed compared with that of other wetlands (i.e. reservoirs, artificial ponds, rivers and watercourses), offering less suitable breeding sites for nesting.

Agricultural intensification and changes in bird communities: winning and losing species in a world of change

Agricultural intensification has had deleterious effects on farmland bird populations at a scale that is comparable to deforestation and global climate change, thus being a major anthropogenic threat to biodiversity (Donald *et al*., 2001). The key to explain this phenomenon seems to be the loss of ecological heterogeneity, which is important in maintaining resources for species-rich communities of organisms (Benton *et al*., 2003). Examples of steppe and pseudo-steppe bird species declining after the intensification of agricultural practices are numerous in Spain, which is the western European stronghold of many of these species (Rodríguez & de Juana, 1991; Tella, Torre & Sánchez, 1996; Suárez *et al*., 1997; Blanco *et al*., 1998; Tella *et al*., 1998). However, there are also some cases of species gaining new habitat opportunities due to the development of intensive crops and the associated agricultural infrastructure. The abundance of some waterbirds at artificial ponds, for example, seems to be larger in comparison with their populations in natural (but degraded) wetlands (Sebastián-González *et al*., 2010). Our results indicate that a raptor species using wetlands for breeding while relying on the surrounding vegetation for feeding can also take advantage of agricultural intensification, at least when food availability is not compromised. Given what is known about other European Marsh harrier populations, many of which have undergone recent expansion (BirdLife International, 2004), our results suggest that this bird could be a winning species in a world of change.

Direct anthropogenic introductions are not the only mechanism favouring the spread of species towards new environments. Human activities are causing many types of alterations in natural landscapes that can be exploited by certain species with positive responses towards these transformations (McKinney & Lockwood, 1999). However, the benefits of transformation on some widespread species (referred to as 'winning species'; McKinney & Lockwood, 1999), such as waterbirds in our study area (Estrada *et al*., 2004), and the harmful effects on others (referred to as 'losing species'; McKinney & Lockwood, 1999), such as pseudo-steppe birds in our study area (Estrada *et al*., 2004), could lead to biotic homogenization processes that can be considered an unprecedented form of global change (Ricciardi, 2007) and one of the most important forms of biological impoverishment world-wide (Olden *et al*., 2004).

Raptors have been proposed as appropriate indicators for monitoring changes at an ecosystem scale because they are at the top of food chains, occurring across a broad gradient of anthropogenic disturbance, from pristine wilderness to metropolitan areas, and because individual species are often associated with particular habitat types (e.g. Tella *et al*., 1998; Sánchez-Zapata *et al*., 2003; Carrete & Donázar, 2005; Carrete *et al*., 2009). Moreover, in some biological systems, these top predators are strongly associated with high biodiversity values (Sergio *et al*., 2008). However, the adaptation of some raptors to human-altered environments (Anderson, 2001; Panasci & Whitacre, 2002; present results) makes their widespread potential use as valuable indicators of natural habitat quality controversial (Rodríguez-Estrella, Donázar & Hiraldo, 1998; Carrete *et al*., 2009). Our results suggest that Marsh harriers respond positively to some degree of intensification, which reduces their usefulness as an indicator species of natural habitat quality. Moreover, preliminary analysis of changes in productivity with respect to the habitat characteristics measured in this study suggests that no significant relationships exist between this demographic parameter and habitat attributes, suggesting that none of the habitats occupied in this area are population sinks. However, further research is needed to determine whether finer cues such as individual survival can indicate other forms of ecosystem degradation such as water pollution or pesticide use during crop development.

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FORAGING HABITAT USE AND SELECTION OF WESTERN MARSH-HARRIERS (*CIRCUS AERUGINOSUS*) IN INTENSIVE AGRICULTURAL LANDSCAPES

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ABSTRACT.—We studied foraging habitat use and selection of the Western Marsh-Harrier (*Circus aeruginosus*) in agricultural areas, by radio-tracking seven individuals in the Ebro Basin (northern Iberian Peninsula). Generalized Linear Mixed Models indicated that the probability that a certain point within the study area was used by harriers was influenced by both distance to the nesting site and habitat. Probability of occurrence decreased with the distance to the nesting site and increased in wetlands and herbaceous crops (both those with high-intensity agricultural practices and those with low-intensity agriculture). In areas dominated by high-intensity herbaceous crops (irrigated maize, cereal, and alfalfa), marsh-harriers ($n = 2$) did not use all crop types equally throughout the year; they selected different crops in relation to the crop stage (growing, fallow, and stubble) and to particular agricultural practices (irrigated, unirrigated).

KEY WORDS: *Western Marsh-Harrier*; Circus aeruginosus; *agriculture*; *habitat use*; *habitat selection*; *radiotelemetry*.

USO Y SELECCIÓN DEL HABITAT DE FORRAJEO DE *CIRCUS AERUGINOSUS* EN PAISAJES DE AGRICULTURA INTENSIVA

RESUMEN.—Estudiamos la selección y el uso del hábitat del aguilucho *Circus aeruginosus* en áreas agrícolas, mediante el uso de radio-telemetría en 7 individuos en la depresión del Ebro (en el norte de la Península Ibérica). La aplicación de Modelos Lineales Generalizados Mixtos mostró que la probabilidad de uso de un punto dentro del área de estudio dependía tanto de la distancia a la zona de cría como del hábitat. La probabilidad de ocurrencia disminuyó con la distancia al nido y aumentó en zonas húmedas y cultivos herbáceos (tanto en campos de uso poco intensivos de secano como en campos de uso muy intensivos de regadío). En zonas dominadas por cultivos herbáceos intensivos (maíz, cereal y alfalfa), los aguilucho ($n = 2$) no utilizaron todos los cultivos por igual a lo largo del año. Además seleccionaron diferentes cultivos en función de su estadio (crecimiento, sembrado, rastrojo) y de determinadas prácticas agrícolas (con y sin riego).

[Traducción del equipo editorial]

Food supply is one of the main limiting factors for raptors (Newton 1979). Breeding densities and breeding success can vary in relation to food availability (Salamolard et al. 2000, Redpath et al. 2002). Land-use change from extensive dry agriculture into intensive farmlands has been described as a major cause leading to impoverished food resources and reduction or degradation of foraging habitats for many open-habitat birds occupying farmlands (Tella et al. 1998, Newton 2004). However, agricultural intensification has not affected all species in the same way, and some species can still take advantage of such modified environments (Siriwardena et al. 1998).

The Western Marsh–Harrier (*Circus aeruginosus*) is a medium-sized raptor that occurs within a wide range of open habitats, including agricultural landscapes. During recent decades, the species has undergone a moderate increase (<10% in Europe; BirdLife International 2004). In some areas, such as the Iberian Peninsula, the increase and spread of the Western Marsh-Harrier population has been noticeable in some agricultural regions (Molina and Martínez 2008). There the species seems to be benefiting from both artificial irrigation ponds and reservoirs for nesting habitat and from the intensive herbaceous crops in their surroundings for hunting (Cardador et al. 2011). However, little is known

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about foraging habitat use and selection of the Western Marsh-Harrier in agricultural areas (Kitowski 2007, Luo et al. 2010). Because male harriers provide most of the food during the nesting period (Simmons 2000), we investigated the foraging habitat use and selection of male marsh-harriers in an agricultural region within the Iberian Peninsula in relation to habitat type and distance to the breeding site.

METHODS

Study Area. The study area was located in the Catalan Ebro Basin, in the northeastern portion of the Iberian Peninsula. The landscape is mostly low-lying and flat, broken by discrete ranges of small hills (0–400 m asl), and has a semiarid Mediterranean climate. During the twentieth century, agriculture in this area underwent several changes, including the replacement of traditionally cultivated cereals (i.e., spring-sown wheat and barley) with a variety of alternative crops (e.g., winter-sown wheat and barley, alfalfa, maize, or fruit trees), the establishment of irrigation systems, the increase in the use of fertilizers and pesticides, and the reduction or disappearance of fallow systems and field margins (Tella et al. 1998). However, the degree of intensification was not homogeneous in the study area, which is now composed of a mixture of high-intensity herbaceous crops (mainly irrigated alfalfa, irrigated winter-sown wheat and barley, and some irrigated maize crops), high-intensity orchards (mainly irrigated peach, pear, and apple trees), low-intensity herbaceous crops (mainly dry winter-sown wheat and barley) and low-intensity orchards (mainly dry almond and olive trees; Aldomà 2004).

We captured harriers and tracked them at two sites, 30 km apart from each other: the ''intensive farmland'' (41°43′N, 0°23′E), which was dominated by high-intensity herbaceous crops and had scattered small ponds where one, or occasionally two, harrier nests were found (Fig. 1); and the ''mixed farmland'' (41°29′N, 0°30′E), which was a more heterogeneous area, centered on Utxesa, a 280-ha reservoir that harbors a colony of 14–16 breeding pairs (Fig. 1).

Capture and Radiotelemetry. During the breeding periods (May–June) 2000–03, we captured and tagged two male harriers in the intensive farmland and five in the mixed farmland. Marsh-harriers were captured in the breeding areas using bal-chatri traps ($n = 6$) or bow nets ($n = 1$). Each harrier was fitted with a backpack transmitter: 12-g AYAMA-Segutel (Mataró, Barcelona, Spain) transmitters or Biotrack (Wareham, Dorset, U.K.) transmitters. Tracking of the birds was conducted from a car. We began each tracking session near the breeding site (<1000 m), and we then followed the signals until the bird was located (i.e., when we knew the position of the bird to within 100 m by visual contact or radio-signal proximity). We used AYAMA or TELEVIT-900 receivers and a three-element Yagi antenna. Tracking sessions lasted on average 6 ± 2 hr ($n = 283$ tracking sessions). Birds were located at all hours between sunrise and sunset. Tagged harriers in the mixed farmland were tracked following a rotational scheme; once one harrier was located, we searched for the next one. Tagged harriers in the intensive farmland could be tracked continuously. We noted the habitat (low intensive herbaceous crops, low intensive fruit trees, high intensive herbaceous crops, high intensive fruit trees, and wetlands; classified in the field by the observer), and plotted each location on 1:25 000 aerial photographs. To avoid autocorrelation, the minimum interval between successive locations of a single bird was 2.5 hr in both the intensive and mixed farmlands and, on average, 2.31 ± 1.02 relocations per harrier per tracking day ($n = 340$ tracking days) were obtained. In cases when several locations were obtained within one 2.5-hr period, the earliest location was selected for inclusion in the dataset. In the intensive farmland (where harriers were tracked continuously), we noted the total time harriers were observed searching for food in fields with different herbaceous crops (alfalfa, cereal and maize), crop stages (growing, fallow and stubble) and agricultural practices (whether the crops were being irrigated or not) during the entire tracking session. We also recorded hunting attempts when witnessed. Harriers were monitored during periods ranging from 1 to 15 mo and were located 1.7 ± 1.4 d/wk ($n = 197$ weeks). We did not expect biases associated with differences in tracking years or periods because marsh-harriers showed high site-fidelity to foraging areas both between and within years in the study area (Cardador et al. 2009).

Habitat Selection. We used a generalized linear mixed model (GLMM; binomial error distribution; logit-link function), to describe mathematically the probability that a certain point within the study area was used by birds in relation to its habitat characteristics (included as a categorical variable with five levels: low intensive herbaceous crops, low intensive fruit trees, high intensive herbaceous crops, high

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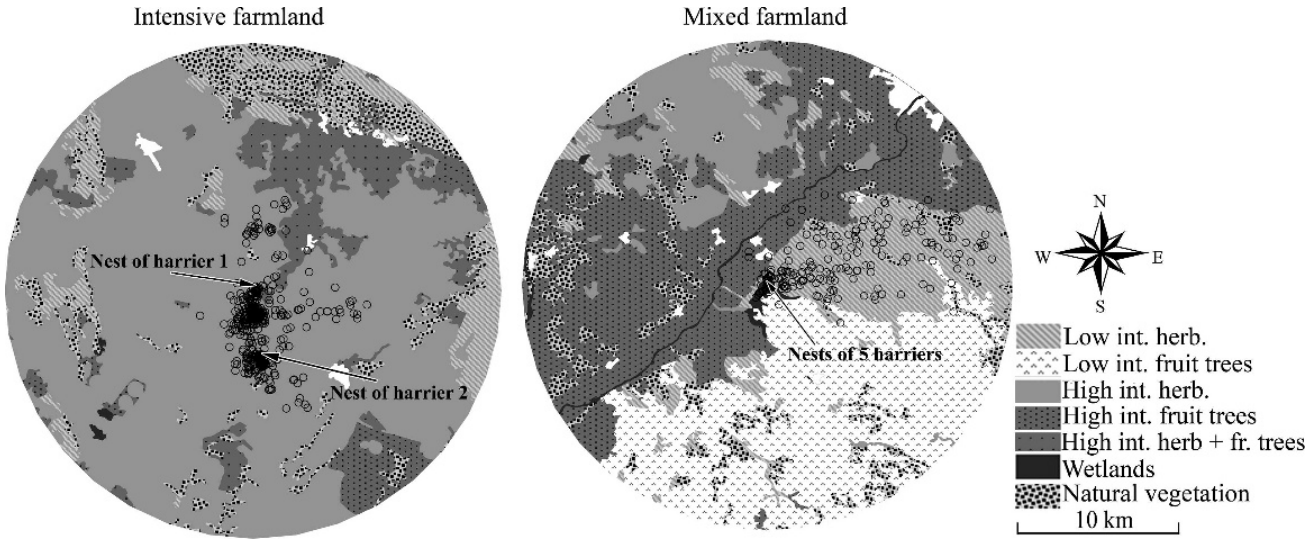


Figure 1. Study area and radio-locations of the seven tagged Western Marsh-Harriers in intensive and mixed farmlands, Catalan Ebro Basin, Spain, May 2000–April 2004. Nest position and available habitats are shown.

intensive fruit trees, and wetlands) and its distance to the nesting site. The dataset included radio-locations of individuals (value 1) and an equal number of randomly selected points not used (value 0) but included within the available study area (Carrete and Donázar 2005). We considered that the total study area was a circle of 15-km radius (the maximum distance from the study site recorded during the study period) focused on the nesting sites. Random points were obtained as independent *x,y* random coordinates within the uniformly sampled study area by using the Animal Movement extension of ArcView 3.2. Habitat types at those points were obtained from a 1:50 000 digital land-use map (Departament de Medi Ambient i Habitatge de la Generalitat de Catalunya, year 1997). In the construction of the model, we included ''individual'' and ''farmland'' as a nested random factor (individual within farmland) in the model to control for the potential nonindependence of data associated with repeated measures of the same individual and farmland.

In the intensive farmland, where harriers ($n = 2$) could be tracked continuously, we also analyzed habitat selection in relation to different herbaceous crops (alfalfa, cereal, and maize), crop stages (growing, fallow, and stubble) and agricultural practices (whether the crops were being irrigated or not). Following Ursúa et al. (2005), habitat selection was analyzed using the Savage selectivity index $w_i = U_i/p_i$, where U_i is the proportion of hunting attempts recorded in a given habitat and p_i is the proportion of total foraging time harriers spent searching for food in that habitat. This index varies

Table 1. Results of GLMM to analyze foraging habitat use of seven male Western Marsh-Harriers relative to habitat and distance to the nesting site (Ebro Basin, 2000–04).

PARAMETER[1]	ESTIMATE	SE	*t*-VALUE	*P*
Intercept	1.5	0.5	3.3	0.017
Distance	−0.0005	0.00003	−19.3	<0.001
Habitat				
Wetlands	2.1	0.6	3.4	<0.001
Low int. herb.	3.5	0.5	7.6	<0.001
High. int. herb.	1.5	0.4	3.4	<0.001
Low int. fruit trees	−0.25	0.9	−0.3	0.78
High int. fruit trees	–	–	–	–

[1] These results were obtained while controlling for the potential effect of individual and farmland fitted as a nested random term (individual within farmland) in the model ($P = 0.35$). Deviance explained by the model: 53.7%; df = 1259.

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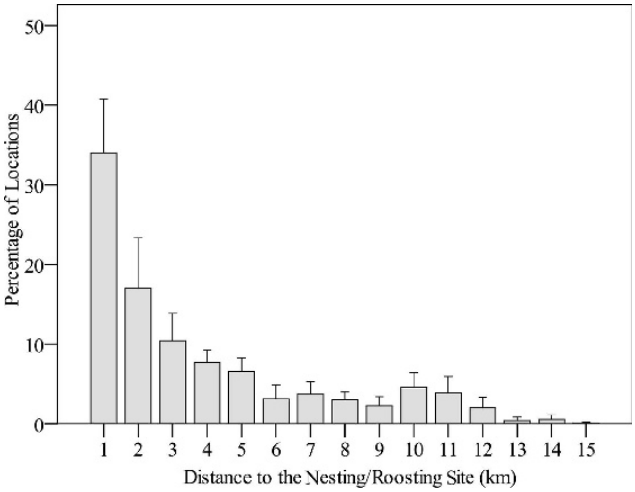


Figure 2. Distribution of distances to the nesting/roosting sites of seven radio-tagged male Western Marsh-Harriers (mean ± SE), relative to distances to the nesting/roosting sites, Catalan Ebro Basin, Spain, May 2000–April 2004.

from 0 (maximum negative selection) to ∞ (maximum positive selection), 1 indicating no selection. The statistical significance of this index was obtained by comparing the statistic $(w_i - 1)^2/\mathrm{se}(w_i)^2$ with the corresponding critical value of a χ^2 distribution with one degree of freedom. The standard error of the index $[\mathrm{se}(w_i)]$ was calculated by $[(1 - p_i)/u_+ * p_i)]^{1/2}$ where u_+ was the total number of hunting attempts sampled.

RESULTS

Seven male harriers were captured and radio-tagged during the study period: five in the mixed farmland (three in 2000 and two in 2001) and two in the intensive farmland (one in 2002 and one in 2003). In total, 678 radio-locations were obtained. The GLMM analysis indicated that the probability that a certain point within the study area was used by harriers was influenced by both distance to the nesting site ($F_{1,1259} = 373$, $P < 0.001$) and habitat ($F_{4,1259} = 22$, $P < 0.001$). Probability of occurrence decreased with the distance to the nesting site and increased in wetlands and herbaceous crops, both high intensive and low intensive (Table 1, Fig. 2).

Radio-tagged harriers in the intensive farmland ($n = 2$) were observed searching for food a total of 227 hr. They principally used alfalfa, maize, and cereal fields (94 ± 2% of total time, $n = 2$). They used alfalfa all year round. Maize use increased during autumn and winter, and cereal use increased during winter and spring (Fig. 3). In total, we observed 455 hunting attempts during this time. Hunting attempts were not distributed at random over the herbaceous crops prospected by harriers. On the contrary, harriers selected different crops relative to habitat type, crop stage, and irrigation (Fig. 4).

DISCUSSION

Raptors are highly mobile predators, which typically have large home ranges, and which often use more than one type of habitat to satisfy their ecological requirements (Newton 1979). As expected, habitat use of marsh-harriers was mainly delineated by two different habitat types: wetlands (which coincided with the breeding/roosting sites and represented <2% of total available habitat), and the herbaceous crops around wetlands (Schipper 1977, Altenburg et al. 1982, Luo et al. 2010), which may offer an adequate vegetation structure for hunting (Preston 1990). Relative to farming practices, marsh-harriers used both low intensive and high intensive herbaceous crops. Proximity to the nesting site also influenced the probability that an area was used for foraging. Although male harriers can hunt far from the nest (up to 14–15 km), most hunting locations were within 3 km of the bird's nest (62 ± 21%, $n = 7$, Fig. 2). Indeed, differences in crop distribution around breeding sites may explain why marsh-harriers in the intensive farmland mainly foraged in high intensive herbaceous crops while those in the mixed farmland mainly foraged in low intensive herbaceous crops (Fig. 1). As in other central-place foragers, the foraging-habitat selection pattern of marsh-harrier may result from a trade-off between the quality of the different patches and the distance at which they were located from the nesting/roosting sites (Carrete and Donázar 2005, Cardador et al. 2009).

Within intensive irrigated farmlands, Western Marsh-Harriers ($n = 2$) mainly foraged in alfalfa fields (81 ± 2% of total foraging time), followed by cereal (9 ± 2%) and maize fields (4 ± 2%). These percentages differed from those estimated from 1:5 000 aerial photographs (Institut Cartogràfic de Catalunya) ground-truthed in the field in 2003 over an area of approximately 90 km² (alfalfa: 50%; cereals: 25%; maize: 15%), suggesting that harriers did not prospect all crops at random. Maize fields, which are sown in April–May, were scarcely used before September, probably because their green cover is too dense and high (0.6–2 m) for hunting and were used primarily during autumn and winter when fields are either covered with stub-

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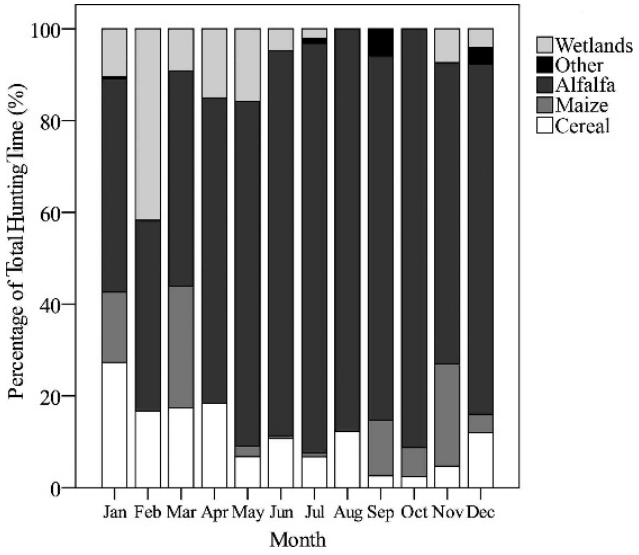


Figure 3. Proportion of hunting time in different crop types by two radio-tagged marsh-harriers in the intensive farmland throughout the year. Mean values for the two harriers are shown. Total observed time = 227 hr.

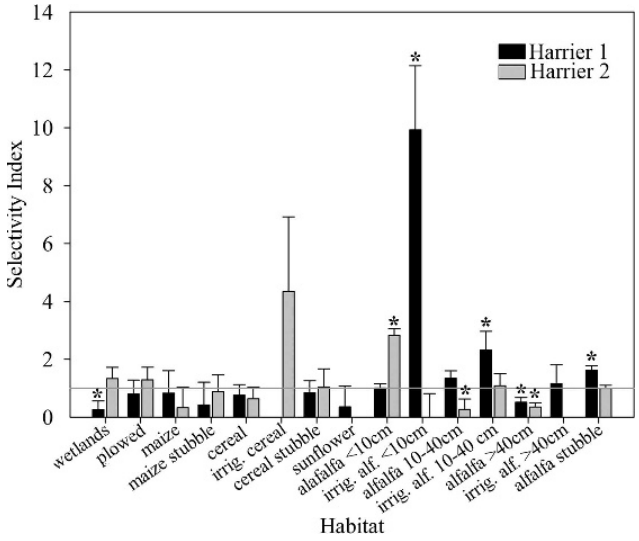


Figure 4. Savage selectivity index (±SE) for each habitat for the two radio-tagged marsh-harriers in the intensive farmland (Catalan Ebro Basin, 2002–04). The horizontal line shows the index value (=1) indicating no habitat selection. Asterisks indicate significant positive/negative selection.

ble or plowed (Ursúa et al. 2005). When harriers used them, hunting attempts were distributed as expected based on the amount of foraging time spent in such crops. Wheat and barley fields were used primarily during winter and spring, during their growing period (before reaching maximum vegetative cover) and harriers did not preferentially use them for foraging. In contrast, Western Marsh-Harriers took advantage of the presence of regularly mown fields (intensively managed alfalfa), which may offer an adequate vegetation structure for hunting during most of the year, especially when crops are <10 cm in height or when stubble is present (Ursúa et al. 2005, Kitowski 2007), and hunting attempts were more frequent than expected based on foraging time spent in these habitats. Interestingly, one of the harriers also increased the number of foraging attempts per hunting-flight time in some fields (alfalfa <10 cm and alfalfa of 10–40 cm) during irrigation. Our observations at the site suggested that the irrigation of such fields may increase the accessibility of small mammals to harriers. The small sample size in our study prevented further analysis of habitat selection and precluded extrapolation of the results outside the studied population.

Although intensification has been considered a major threat for this and other open-habitat bird species occupying farmlands (Benton et al. 2003), our results agreed with previous findings, and showed that intensive agriculture still may offer adequate foraging habitats for some species, at least when food availability is not compromised (Ursúa et al. 2005, Balbontín et al. 2008). Like other generalist predators, Western Marsh-Harriers may be able to exploit transient resources in intensive agricultural habitats, and may be less sensitive to habitat degradation than other more specialized species (Siriwardena et al. 1998).

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Feeding behaviour and diet composition of Marsh Harriers *Circus aeruginosus* in agricultural landscapes

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Bird Study (2012) **59**, 228–235

Feeding behaviour and diet composition of Marsh Harriers *Circus aeruginosus* in agricultural landscapes

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Capsule Diet composition and feeding behaviour of breeding birds differed between areas of high and low agricultural intensification.
Aims To analyze diet composition and feeding behaviour of Marsh Harriers *Circus aeruginosus* in agricultural landscapes with different degrees of agricultural intensification.
Methods Diet composition and feeding behaviour was assessed by focal observations at nests during breeding seasons in 2001–03 in two agricultural areas within the eastern Ebro Basin (on the northern Iberian Peninsula) which differed in their degree of agricultural intensification: a traditional non-irrigated and non-intensively managed farmland area; and an irrigated and intensively managed farmland area.
Results Marsh Harriers in the intensively managed area took higher percentages of small mammals than harriers in the traditional area managed at low-intensity, where they also had a more diverse diet. Male delivery rates increased throughout the breeding season in the intensively managed area. On the contrary, no significant trend in delivery rates was observed in the traditional area, where harriers turned to larger prey late in the season.
Conclusion Marsh Harriers probably took advantage of different prey types in intensively managed and traditional farmland according to prey availability. Differences in diet composition seem to be related to different foraging strategies adopted by harriers in traditional and intensively managed farmlands. In the traditionally managed farmlands, harriers responded to increases in the energy requirements of the brood over the course of the breeding season by taking larger prey, whereas in the intensively managed farmlands harriers increased feeding rates.

Agricultural intensification over the last six decades has been linked to decreased biodiversity and declines in breeding populations of several farmland bird species (Donald *et al.* 2001). This phenomenon is partially explained by the loss of ecological heterogeneity, which is important in maintaining resources for species-rich communities (Benton *et al.* 2003, Fahrig *et al.* 2011). Among other processes, land-use change from extensive traditional agriculture into intensively managed farmland has led to changes in food availability and reduction or degradation of foraging habitats for many bird species (Newton 2004). However, agricultural intensification has not affected all species in the same way and transformed landscapes still provide suitable habitats for species tolerant of anthropogenic alterations (Siriwardena *et al.* 1998).

Food supply is one of the main limiting factors for birds of prey. Breeding densities and breeding parameters such as laying date, clutch size or productivity can vary in relation to food availability or diet composition (Newton 1979, Redpath *et al.* 2002, Arroyo & García 2006). Food availability is mainly determined by both the abundance and accessibility of food resources (Preston 1990), which in turn can be affected by changes in the composition and structure of foraging habitats. Therefore, the effective conservation and management of birds of prey in anthropogenic landscapes such as farmland requires an understanding of their trophic ecology (Butet & Leroux 2001, Amar *et al.* 2003).

Marsh Harriers *Circus aeruginosus* occur within a wide range of habitats, including farmland. After a dramatic

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population decline during 1960–80, the European population has shown a moderate increase (< 10%) in recent decades (Cramp & Simmons 1994, BirdLife International 2004). Unlike many other farmland birds, this species has made a notable recovery in some agricultural areas, including those dominated by intensively managed crops. Within some areas such as the Iberian Peninsula, the increase in the Marsh Harrier population has been accompanied by increases in both its geographic range and in the variety of habitats it occupies, with some pairs nesting in artificial irrigation ponds and reservoirs related to intensive agricultural practices (Cardador *et al.* 2011). However, diet composition and feeding behaviour of Marsh Harriers in these new habitats remain unknown.

We investigated whether diet composition and food provisioning patterns of Marsh Harriers in an expanding Spanish population differed between two localities with different degrees of agricultural intensification. We also surveyed small mammal abundances to assess whether possible variations in diet and feeding behaviour were related to food abundance.

METHODS

Study area

The study area was located within the eastern Ebro Basin on the northern Iberian Peninsula. The area is mostly low-lying and flat, broken by discrete ranges of small hills (0–400 m asl) and with a semi-arid Mediterranean climate. Almost one-third is covered by farmland, which is managed at a wide range of intensities. Diet composition and feeding behaviour of Marsh Harriers were monitored at two sites, 30 km apart. The 'intensively managed farmland' site (41°43′N, 0°23′E) was dominated by intensively managed herbaceous crops (irrigated winter-sown wheat and barley, irrigated alfalfa and some irrigated maize crops) and had scattered small ponds where one or occasionally two harrier nests were found. The 'traditional farmland' site (41°29′N, 0°30′E) was a more heterogeneous area, centred on Utxesa, a 280-ha reservoir that harboured a colony of 14–16 breeding pairs (Cardador & Mañosa 2011). This area included a large central patch of traditional herbaceous crops managed at low intensity (mostly non-irrigated winter-sown wheat and barley) surrounded by fruit trees. Previous information showed that harriers nesting in this region used the traditional herbaceous crops for hunting (Cardador *et al.* 2009, Cardador & Mañosa 2011).

Data collection

Data were recorded by means of focal observations at nests during breeding seasons (April–August) in 2001–03. We conducted 1049 hours of observation (523 in the traditional farmland site and 526 in the intensively managed farmland site) and recorded 752 prey deliveries to nests from a total of 22 nests (located at Utxesa reservoir) over the 3 years in the traditional farmland site and a total of 10 nests (located in 3 irrigation ponds) in the intensively managed farmland site. Watches were made from vantage points located 200–700 m from the nests using 10 × 42 binoculars and 20–60× telescopes. Observations were conducted between 07:00 and 21:00 hours and lasted on average 4.6 hours ± 2.4 sd ($n = 226$ sessions). For each monitored pair, number of prey deliveries to nests by males and females were recorded. Feeding rates were then calculated for three periods (incubation; nestling; and post-fledging) as the total number of prey delivered to the nest by each sex divided by the total hours of observation (prey h^{-1}), when at least one delivery was recorded. When possible, prey items were identified to class or a lower taxonomic level, and assigned to one of four size categories: very large (e.g. lagomorphs or Mallards *Anas platyrhynchos*); large (e.g., rats *Rattus* spp. or Red-legged Partridges *Alectoris rufa*); medium (e.g., rodents or passerines); and small (e.g. shrews or orthoptera).

Food abundance

An index of abundance of small mammals was recorded in June 2001. In each site, four 1 × 1-km squares located in representative habitat and separated by at least 1 km were delimited. In each square we placed 100 Sherman traps at a minimum spacing of 10 m throughout 2 successive nights (between 21:00 and 08:00 hours approximately). The index of abundance in each square was calculated as the sum of small mammals trapped during the two nights. Sherman traps were not efficient for Mediterranean Pine Voles *Microtus duodecimcostatus*, because they were placed on the ground and voles are mainly fossorial (Torre *et al.* 2010). Therefore, we estimated vole abundance by using an indirect index based on counts of occupied burrows, which we identified as those with fresh digging (Liro 1974). These counts were carried out in six sampling squares of 30 × 30 m within each study site in July 2002, located in representative habitat and separated from one another by at least 170 m.

Data analyses

Identified prey were grouped in five categories: small mammals (i.e. rodents and shrews); large mammals (i.e. lagomorphs and carnivores); birds; reptiles; and fish. For each site and sex, we calculated diet diversity using the Simpson inverse diversity index, which tends to weight in favour of abundant prey types. We used sample-based random re-sampling (50 runs) to derive estimates of prey diversity that are independent of sample size or completeness by using ESTIMATES 8.2 (Colwell 1999). Differences in dietary diversity between groups were established by visually comparing sample-based accumulation curves and their corresponding 95% CIs.

We then analyzed differences in diet composition between sites and whether those differences varied between sexes or periods of the breeding season. For these analyses, we pooled into single categories small mammals and other mammals ('mammals') and those categories of prey that were very rarely taken by Marsh Harriers ('other prey' – i.e. reptiles and fishes; see Fig. 1). Then, for each prey category considered (i.e. mammals, birds and other prey), we conducted a GLMM with site, sex, period (fitted as an ordinal variable, with incubation < nestling < post-fledging) and the interactions between site and period and between site and sex as fixed factors. We used a hierarchical backward stepwise elimination, following Crawley (1993) – all explanatory variables were fitted to the model, extracted one at a time from this maximal model (starting from higher-order interactions), and the associated change in model deviance was assessed by an *F*-test. Main effects contained in a significant interaction term were retained in models. We fitted a binomial error distribution for the dependent variables (the probability that the delivered item was or was not the category of prey considered in each model), and used a logit-link function. We included nest and year as random terms in those models to control for repeated observations of the same nests and years.

We tested for differences in feeding rates between sites and whether those differences varied among sexes or periods of the breeding cycle by using GLMMs with a hierarchical backward elimination method (see earlier). We fitted period, site, sex and the interactions between site and sex and between site and period as fixed factors in those models. We used the normal error distribution and the identity-link function for continuous data (i.e. feeding rates: prey h^{-1}) and the binomial error

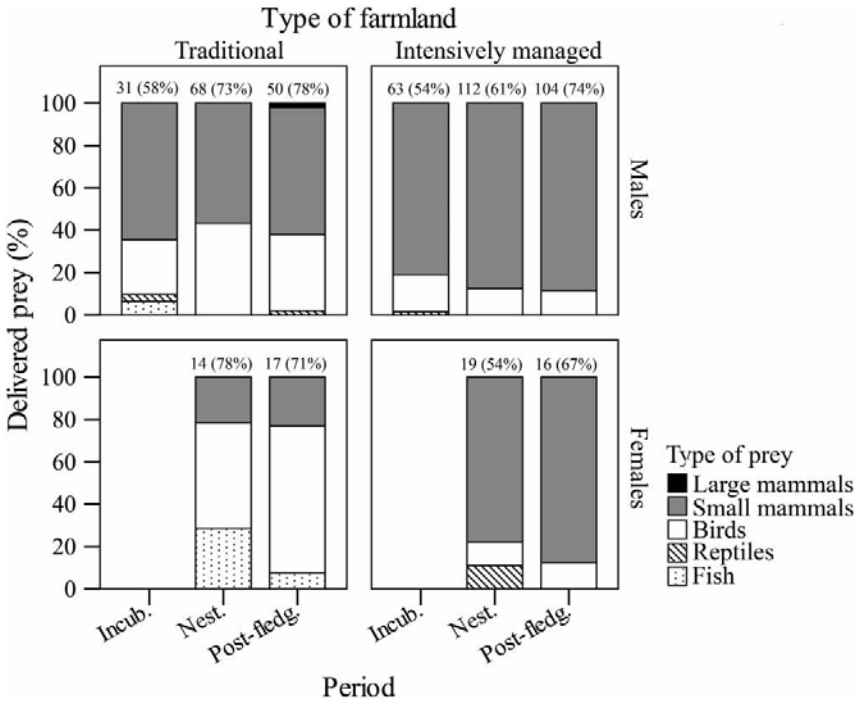


Figure 1. Comparative data on diet composition according to agricultural landscapes (traditional or intensively managed), period of the breeding cycle, and sexes of Marsh Harriers in the Catalan Ebro Basin (Spain), 2001–03.
Number of identified prey and percentage that it represents from total observed deliveries (in parentheses) are shown.

distribution and the logit-link function for presence/absence data (i.e. the probability that the delivered item was or was not the category of prey size considered – large, medium or small; very large prey were not considered for analyses because they were very rarely taken by the species, < 1% of total identify prey). We included nest and year as random terms in those models to control for non-independence of data.

Finally, we tested for differences in food abundance between sites by using GLMS with site as a categorical factor. We fitted a Poisson error distribution for the dependent variables (abundance of each category of food) and used a log-link function. All tests were performed using SAS 9.2.

RESULTS

Diet composition

Marsh Harriers occupying the traditional site had a more diverse diet than harriers occupying the intensively managed site (Fig. 2). Delivery rates of mammals and birds to the nest by monitored harriers significantly differed between the type of farmland (Table 1) and those differences seemed consistent among different periods of the breeding season (variable period not retained in models). Marsh Harriers in the traditional site delivered fewer mammals and more birds than in the intensively managed one (Fig. 1, Table 1). Males delivered more mammals to the nest than females (Fig. 1, Table 1), but differences between sexes were more pronounced in the traditional farmland site (significant interaction 'site × sex'). Mediterranean

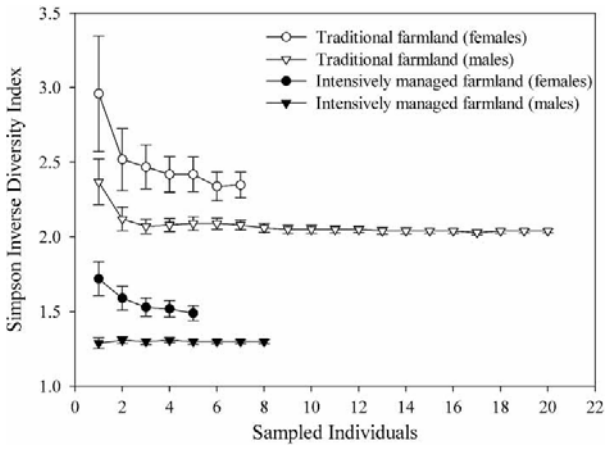


Figure 2. Sample-based accumulation curves for Simpson inverse diet diversity index of male and female Marsh Harriers occupying traditional and intensively managed farmlands.

Table 1. Results from GLMMS for differences in the consumption of different prey categories between the two studied sites (traditional versus intensively managed farmlands), periods of the breeding cycle and sexes.

Parameter	Estimate ± se	df	*F*-value	*P*-value
Mammals				
Traditional	−2.80 ± 0.63	464	26.08	< 0.001
Males	0.33 ± 0.46	464	9.01	0.003
Traditional × Males	1.40 ± 0.70	464	3.94	0.047
Birds				
Traditional	1.35 ± 0.35	466	12.97	< 0.001
Other prey				
Traditional	1.97 ± 0.83	464	5.09	0.024
Males	−3.73 ± 0.81	464	23.60	< 0.001
Period	−1.17 ± 0.51	464	5.40	0.021

Pine voles were the most commonly consumed prey within the small mammal category in the intensively managed site. Conversely, mice (*Mus* sp.) were the most commonly consumed prey in the traditional site (Fig. 3). Other prey accounted only for a small proportion of prey items in both farmland types (Fig. 1), but differences between sites, sexes and periods of the breeding season were found (Table 1). Deliveries of other prey were more frequent in the traditional farmland site, decreased late in the season, and were conducted more frequently by females than by males.

Abundance indices of small mammals differed between sites. Abundance indices of Algerian Mice *Mus spretus* ($\chi^2 = 25, P < 0.001$) and Wood Mice *Apodemus sylvaticus* ($\chi^2 = 4.16, P = 0.04$) were higher in the traditional site, while abundance indices of voles (voles' burrows: $\chi^2 = 216, P < 0.001$) and Greater White-toothed Shrews *Crocidura russula* ($\chi^2 = 4.82, P = 0.03$) were higher in the intensively managed site (Fig. 4).

Variation of feeding rates

A significant interaction between site and period and between site and sex was found (Table 2), meaning that the relationship between feeding rates and type of farmland was complex and linked to periods of the breeding cycle and sexes. Feeding rates in the intensively managed farmland site increased throughout the breeding season, while feeding rates remained nearly unchanged in the traditional farmland site (Table 2). Females only provided food to the chicks during nestling and post-fledging periods (Fig. 5) and at a lower rate than males in both traditional and intensively managed farmland, but differences between sexes were less pronounced in the traditional farmland site (significant interaction 'site × sex'; Table 2).

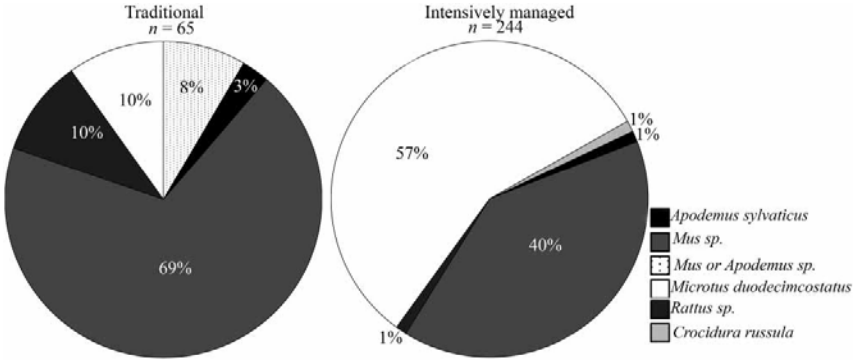


Figure 3. Occurrence of different prey species, within the 'small mammal' category, in traditional and intensively managed farmland sites. Number of total small mammals that could be identified to genus or species is shown; the total number of small mammal deliveries observed was 94 in the traditional farmland site and 264 in the intensively managed farmland site.

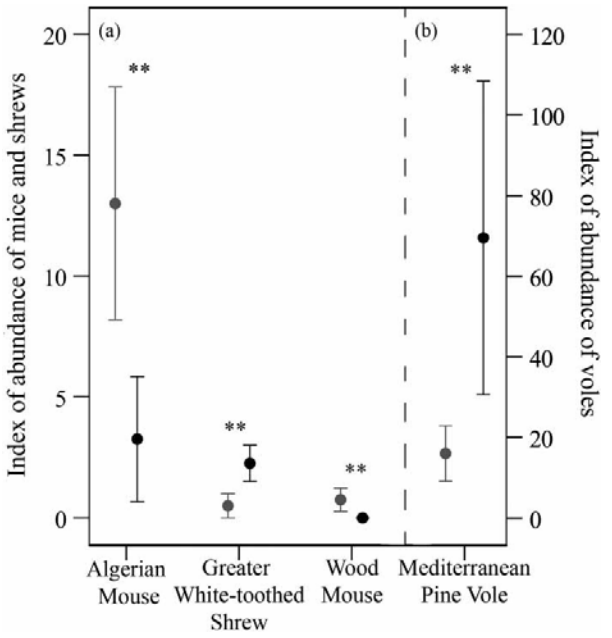


Figure 4. Variation in the abundance (mean ± se) of small mammals between traditional (•) and intensified (•) farmlands in the study area, Abundance was estimated (a) directly, as the number of individuals caught in two successive nights by trapping in 1 × 1-km squares or (b) indirectly, as the number of active burrows counted within 30 × 30-m squares.
**Significant differences ($P < 0.005$) in abundances between farmlands based on GLMS (Poisson error distribution, log-link function).

Variation in prey size

Harriers from intensively managed and traditional farmland sites did not take prey of different sizes with the same frequencies throughout the breeding season (significant interactions 'site × period' in final models for medium and large prey; Table 3). Harriers in the traditional farmland site increased the percentage of large prey throughout the breeding season, while harriers in the intensively managed farmland site took an increasing proportion of medium-sized prey as the season progressed (Fig. 6, Table 3). No significant differences were found in the percentage of different prey size categories delivered to the nest between sexes (variable 'sex' not retained in models). No temporal trend was observed in either traditional or intensively managed farmland areas for small prey (none of the variables considered were retained in models).

Table 2. Results from GLMMS for differences in the consumption of large and medium-sized prey between the two studied sites (traditional versus intensively managed farmlands), periods of the breeding cycle and sexes.

Parameter	Estimate ± se	df	*F*-value	*P*-value
Large prey				
Traditional	−1.07 ± 1.07	582	1.00	0.32
Period	−0.32 ± 0.23	582	0.06	0.81
Traditional × Period	0.72 ± 0.33	582	4.66	0.03
Medium prey				
Traditional	−1.60 ± 0.88	582	3.32	0.07
Period	0.38 ± 0.16	582	0.10	0.75
Traditional × Period	−0.68 ± 0.27	582	6.26	0.01

Table 3. Results from GLMMS for differences in feeding rates between the two studied sites (traditional versus intensively managed farmlands), periods of the breeding cycle and sexes.

Parameter	Estimate ± se	df	*F*-value	*P*-value
Traditional	0.60 ± 0.23	46	4.52	0.039
Males	0.52 ± 0.09	46	41.37	< 0.001
Period	0.18 ± 0.05	46	11.75	0.001
Traditional × Period	−0.16 ± 0.06	46	7.53	0.010
Traditional × Males	−0.37 ± 0.10	46	12.41	0.001

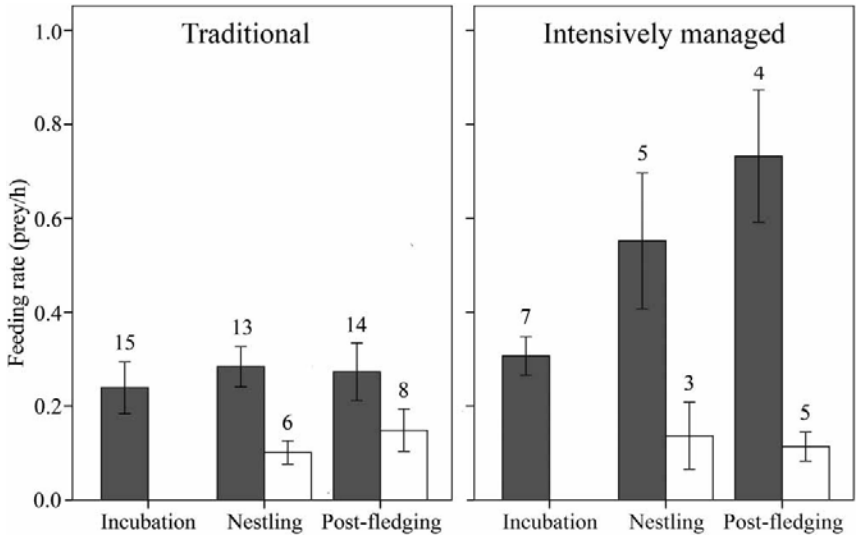


Figure 5. Mean ± se food delivery rates (prey h^{-1}) for male (grey bars) and female (open bars) Marsh Harriers throughout the breeding season in traditional and intensified farmlands, within the Catalan Ebro basin (Spain), 2001–03.
Sample sizes (number of harriers observed delivering food at least once in each period) are shown above the histogram bars.

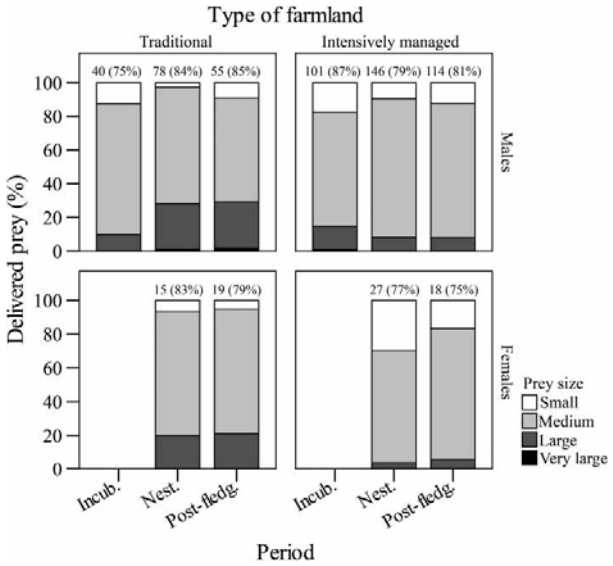


Figure 6. Seasonal variation in diet composition according to prey size delivered by male and female Marsh Harriers breeding in traditional and intensively managed farmlands.
Number of total identified prey and percentage that it represents of the total observed deliveries (in parentheses) are shown.

DISCUSSION

The diet of Marsh Harriers in our study area mainly comprised mammals and birds, with other groups, such as reptiles or fishes, accounting for a much lower proportion of prey items. These results agree with those reported for other European populations (Cramp & Simmons 1994), indicating consistency in the diet of Marsh Harriers over much of their geographic range. Within mammals, small mammals were the most commonly consumed prey, contrasting with previous studies in central Spain, where Marsh Harriers mainly preyed on lagomorphs (González 1991). Marsh Harriers in intensively managed areas took larger percentages of small mammals and had a less diverse diet than those in more traditional habitats. Additionally, Marsh Harriers in traditional and intensively managed farmlands did not prey on the same small mammals species. In intensively managed farmland, Mediterranean Pine Voles were the most frequently taken species within the 'small mammals' category (approximately 60%); but, they only comprised 10% in traditional farmland, where mouse species were the most commonly consumed prey within this category. Although limited (only one year), censuses of small mammals in the study area suggest that diet composition of Marsh Harriers reflected differences in the abundance of such prey between farmlands. This result seems consistent with previous studies, which have shown that microtine species reach higher densities than other small rodents species in more intensified agricultural systems (Millán de la Peña *et al.* 2003, Burel *et al.* 2004). However, we cannot completely discard the possibility that other local or temporal processes, not related to intensification, could be affecting our results.

Optimal foraging theory predicts that individuals will forage in such a way as to maximize their energy gain and that diet diversity will decrease as abundance of preferred prey increases (Pyke *et al.* 1977). Small mammals have been described as a preferred and high-quality prey for *Circus* species in some regions, positively affecting breeding parameters and body condition (Butet & Leroux 1993 , Dijkstra & Zijlstra 1997, Sternalski *et al.*2010). Under this perspective it seems that the diet of Marsh Harriers in intensively managed farmland (with a higher proportion of small mammals and less diverse than in traditional farmlands) could be the result of the particularly favourable foraging conditions offered by this habitat in terms of optimal prey availability, as has been suggested by previous studies on ranging behaviour and habitat use (Cardador *et al.* 2011, Cardador & Mañosa 2011). However, this is not the only possible interpretation, as a high percentage of small mammals in the diet of Marsh Harriers in intensively managed habitats could also be a by-product of there being relatively few alternative prey, such as birds (Terraube & Arroyo 2011). More specific studies on foraging efficiency and time budgets in different habitats with prey types are needed to clarify whether small mammals or birds have higher or equal value for Marsh Harriers (Terraube *et al.* 2010) and whether the higher consumption of small mammals in intensively managed farmland is the outcome of an optimal foraging strategy or the result of a reduction of alternative prey diversity and availability.

Differences in diet composition between traditional and intensively managed farmlands in our study area may explain the differences in feeding strategies followed by Marsh Harriers in these two types of landscapes throughout the breeding cycle. Among harriers, males provide most prey items for chicks and females during the breeding season (Cramp & Simmons 1994), while females only help with chick provisioning late in the season, and at lower rates than males, since they need to combine their hunting activities with nest defence (García & Arroyo 2005). In intensively managed areas, where male harriers preyed mainly on small mammals, they increased their feeding rate late in the season (during nestling and post-fledging stage), probably in response to increased energy demands (Underhill-Day 1993, García & Arroyo 2005). On the contrary, in traditional farmlands, where male harriers fed on more diverse prey, they followed a different strategy: they turned to larger prey late in the season, but did not vary their feeding rate.

Agricultural intensification has led to changes in prey composition and abundance in farmland areas (Donald *et al.* 2001, Benton *et al.* 2003). Marsh Harriers in our study area appear to adapt their feeding behaviour to prevailing feeding conditions. Preliminary analysis suggests that mean productivity of birds breeding in intensively managed farmland is similar to that experienced by birds breeding in traditional farmland habitats (Cardador *et al.* 2011). However, the effects of agricultural intensification and related dietary shifts on other demographic parameters such as inter-annual variations in productivity or survival are unknown, and more specific studies will be needed before the global and long-term effects of agricultural intensification on Marsh Harrier population can be elucidated.

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Natural England Commissioned Report NECR207

Functional linkage: How areas that are functionally linked to European sites have been considered when they may be affected by plans and projects - a review of authoritative decisions

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NATURAL ENGLAND

Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Background

The implementation of a wide range of plans or projects can affect species or habitats on sites which have been designated for their nature conservation importance as European Protected Areas.

Areas of land or sea outside of the boundary of a European site may be important ecologically in supporting the populations for which the site has been designated or classified. Occasionally impacts to such habitats can have a significant effect upon the species interest of such sites, where these habitats are considered to be functionally linked to the site.

Natural England advisers need to judge whether the impacts on habitats that are functionally linked to a site, may adversely affect its integrity and whether the effects are significant in light of the conservation objectives for the site.

This report aims to provide an analysis of authoritative decisions that have considered functional linkages in cases assessed under the Habitats Directive and Regulations. It will be used as a referencing tool for Natural England and other decision makers, in particular Natural England advisers involved in casework.

This report should be cited as:

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Further information
This report can be downloaded from the Natural England website: **www.gov.uk/government/organisations/natural-england**. For information on Natural England publications contact the Natural England Enquiry Service on 0845 600 3078 or e-mail **enquiries@naturalengland.org.uk**.

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Summary

Introduction

This report aims to provide an analysis of authoritative decisions which considered effects from projects being considered for authorisation on areas of land or sea that were considered to be functionally linked to a European site, but which lay outside the boundaries of the site. It is intended to serve as a referencing tool for use by Natural England to inform a review of its approach to casework in light of interpretations of the Habitats Directive and Regulations.

In the context of this report, the term 'functional linkage' refers to the role or 'function' that land or sea beyond the boundary of a European site might fulfil in terms of ecologically supporting the populations for which the site was designated or classified. Such land is therefore 'linked' to the European site in question because it provides an important role in maintaining or restoring the population of qualifying species at favourable conservation status. This report only looks at areas which are functionally linked for species rather than qualifying habitats.

An 'authoritative decision' is a decision which has been subject to sufficient scrutiny, at an appropriate level, to impart a degree of *authority*. In this report, 'authoritative decisions' used are those relating to domestic court judgments, and Secretary of State and certain Planning Inspector decisions in respect of a proposed plan or project.

It may be necessary to consider the date of a decision or the extent to which a particular case is consistent with previous judgments or practice before relying upon it in a decision-making process. It is the responsibility of the reader to interpret and apply the findings in this report appropriately. The findings and conclusions of the report should be considered fairly, as a whole, and not quoted, used or applied selectively, in order to support a pre-determined or preferred conclusion.

In Habitats Regulations Assessment the concept of functional linkage is relevant to both the stage 1 'screening decision' and the stage 2 'integrity test'. If effects on functionally linked land or sea are likely to have a significant effect on the population of species for which a European site was designated or classified, those effects must be considered fully in a Habitats Regulations Assessment.

Methodology

The researchers compiled a list of potentially relevant cases drawn from:

i. their own library of decisions, and their empirical knowledge of case work;
ii. a further web-based search of European Court judgments and opinions;
iii. a web-based search for decisions relating to nationally significant infrastructure projects and projects consented under the Electricity and Pipeline Acts in England and Wales and their territorial and UK offshore waters; and

iv. suggestions made by officers in Natural England following an e-mail enquiry of case officers by the research project manager.

Over 180 cases were originally identified as being relevant to the assessment of plans and projects affecting European sites. Following an initial screening exercise, twenty five of these decisions were subject to detailed examination in this review as being relevant to the consideration of functional linkage.

One Court of Appeal case is included only to contribute to the discussion on evidence requirements. Of the twenty four cases subject to further analysis, 19 referred to effects on birds, 4 were related to effects on bats, and two referred to effects on Atlantic salmon. One of the 19 SPA cases also identified potential effects in respect of marine mammals.

Discussion and conclusions

Bats

The issues in the four cases relating to bats examined the potential loss, interruption, or diminution of the ecological value of the routes (flyways) used by the bats from the SAC to reach their foraging grounds, which were spread around the countryside beyond the SAC boundary. Hence the bats would be indirectly affected by way of loss of habitat, or by the interruption or severance of the flyways or by the introduction of deterrent effects in the flyways and/or in the foraging areas. Reduction in ecological value of the foraging areas and/or impediments to the bats reaching their foraging areas could undermine achievement of the conservation objectives of the SACs and therefore affect the conservation status of the bats in the SACs. The failure of the developer in one case to carry out the surveys reasonably required to establish the importance of an area reasonably likely to be part of a critical flyway, led to the refusal of the application and dismissal of the appeal. This was because an appropriate assessment could not be properly completed without it. In all cases the risk to the population of bats, for which the SAC had been designated, arising out of effects which could occur beyond the boundary of the SAC was accepted by the decision maker.

Birds

Twelve SPA cases related to terrestrial or coastal habitats involving a range of waterfowl. Seven related to the marine environment and sea birds. In all cases the decision maker recognised the potential importance of functionally linked land or sea and that it should be treated as part of the Habitats Regulations Assessment.

In the terrestrial or coastal environments, three points became clear from the research:

a) the scope for SPA bird species to use land, whether in close proximity or further away from the SPA, is often limited by urban development, land use patterns, noisy or other disturbing activities or operations, barriers and of course suitability of the habitat, even where there is open land or water;
b) there are often good quality, pre-existing records, such as Wetland Bird Census data, to indicate the use of specifically defined areas outside the SPA by birds;

c) surveys of use, or potential use, of land or water bodies by relevant SPA species is usually reasonably obtainable even if surveys are required over a period of time.

In the terrestrial or coastal environments, the functionally linked land was identified by surveys of actual or probable use, rather than mere assumption that birds from the local SPA might use the area because it might be suitable.

Consequently, in the terrestrial and coastal environments, the possibility of the presence of functionally linked land is more readily identifiable, and the land areas more easily defined as relatively discrete areas, than in the marine environment discussed below. Where pre-existing records were not available, but an area affected by a development appeared likely to be used as functionally linked land, new survey work was undertaken to establish the level of use in order to inform a Habitats Regulations Assessment, screening or appropriate assessment, as the case may have been. Even where good data about levels of use were available, survey effort was continued in some cases to improve understanding.

By contrast, the seven sea bird SPA cases, relating to offshore wind farms, had to approach the potential of functionally linked sea areas differently. All new proposals involved surveys to establish the use of the proposed wind farm areas by all species of birds. It was then necessary to consider whether the birds that were recorded in these offshore areas may reasonably be assumed to be individuals associated with an SPA. This depended in part on the proportion of that species which bred in SPAs, and the distance that the area lay from SPAs for the relevant species. The critical distance was usually the species-specific, maximum recorded foraging distance, or in some cases the known flight paths, which varied considerably from one species to another. No standard cut off distance from an SPA could be used as a surrogate for the risk of a significant effect. All of this was relevant before the ornithological analyses attempted to calculate collision risk or displacement for the birds that were recorded as using the development area.

On the scientific evidence in the Habitats Regulations Assessments accompanying the decision in each of these cases, the calculations of displacement and collision risk modelling related to bird populations which could reasonably be assumed to be those relating to an SPA; in all cases the SPAs potentially affected were specifically identified.

Atlantic salmon

One case concerned effects on individuals of Atlantic salmon, when they would be upstream of the SAC boundary. With the project being located upstream of the designated SAC, the functional linkage between the population for which the SAC had been designated and the individuals potentially affected by the proposed development was clear, because they would all have had to migrate through the SAC to reach the upstream stretches. These risks had not been adequately assessed when the original permission had been granted. In the other case it was the risk to migrating smolt caused by the deterrent effect of noise from piling operations that had the potential to prevent Atlantic salmon from making their seaward migration from the SAC out into the Irish Sea. The risk was avoided by a seasonal restriction on the driving of the piles.

Harbour porpoise and seals

In this case the functional linkage of a wind farm site to 26 pSCIs around the North Sea (most in the territorial seas of other member states), which had been designated for harbour porpoise and two for grey seals and harbour (common) seals was necessarily based on certain assumptions. For each species, the long distances from the European sites and the extensive range over which the species were known to forage led to the conclusion that displacement was not considered to represent a threat to the integrity of any of the sites potentially affected.

Evidence requirements

The 'Boggis' case helps to establish some principles about the requirement for at least credible evidence that there is a functional link between an area that may be affected by development and a European site. In 15 of the 24 other cases examined in detail the researchers considered that there was a good level of survey or other evidence demonstrating a relatively clear (or even obvious) link to the SAC/SPA and its species.

Other cases, in the marine environment, had to be based on reasonable scientific assumptions. But these cases should not be regarded as having a weak evidence base for the links. In the precautionary approach of the Habitats Regulations sufficient evidence pointed to a possibility or a risk of an effect on SPA or SAC populations.

In respect of two cases, the evidence base was considered to be 'poor' with links not well established. In one of these cases a legal challenge to a plan consequently failed. In the other case, referred to under bats above, the developer declined to carry out surveys that could have demonstrated how important the functional link was and how effective mitigation measures might have been, so the application of the precautionary principle led to the refusal of the application and dismissal of the appeal.

In essence, the research shows, amongst other things, that:

a) The identification of an area as functionally linked land in the terrestrial or coastal environment is generally relatively straightforward and readily recognised, but may sometimes not be apparent and may require some initial survey and analysis or collation of pre-existing data, to establish the link;
b) The identification of an area as functionally linked sea is more challenging and has to be approached differently for marine developments; nevertheless an approach in respect of sea birds and marine mammals appears to be developing and although necessarily relying to a greater extent on assumptions, it provides a robust approach which is suitably precautionary without being onerous;
c) Once identified as functionally linked land or sea, the evidence required by decision makers in stages 1 and 2 of the Habitats Regulations Assessment process are no different to those that might reasonably be expected in relation to direct or on-site effects on the European site. The precautionary principle applies equally to functionally linked land and sea. Where effects might be significant and there is insufficient information to ascertain that there would not be an adverse effect on the integrity of a site, in terms of the population of the species for which the site has been

classified or designated, authorisation has been denied; consistently with the provisions of the Regulations. However, in the majority of cases sufficient evidence was available for the decision-maker to conclude that there would be no significant effect, or no adverse effect on site integrity, if the project was authorised.

A Background to this research

A.1 Status of this report

This report sits within a series reviewing the findings of 'authoritative decisions'. It is concerned with how areas of land or sea that are functionally linked to a European site have been considered by authoritative decision makers when projects may affect them. At the time of writing, two other reports are available regarding 'small scale effects' and 'longevity of effects'.

A.2 Who the report is for

The research was commissioned by Natural England "*for the production of a report which can act as a referencing tool for use by Natural England to inform a review of its approach to casework in light of recent interpretations of the Habitats Directive and Regulations"*. Whilst the report has primarily been drafted for Natural England, it will be of interest to all practitioners and advisers working in the assessment of plans and projects under the 'Habitats Regulations'[1].

A.3 Aims of this report

Natural England advisers in casework frequently issue advice on the potential effects that proposed plans or projects might have on European sites. For the purpose of this report the term 'European site' includes:

- Special Protection Areas (SPAs) classified under the EU Birds Directive[2];
- Special Areas of Conservation (SACs) designated under the EU Habitats Directive[3];
- Ramsar Sites listed under the Ramsar Convention[4].

Cases involving proposed SPAs or SACs could also be relevant because of European Court rulings as to how member states should secure the protection of such sites before they are fully designated or classified. Later in this report there are references to 'Sites of Community Importance' or 'SCI', because this is a term widely used in respect of European sites by the European Court and the European Commission.

Advice is given by Natural England based on the best available information in light of the characteristics and specific environmental conditions at the site concerned[5]. However, it can be difficult to ascertain what is acceptable under the specific tests set out in the assessment provisions of the Habitats Regulations (regulation 61), commonly referred to as a 'Habitats Regulations Assessment' or 'HRA'.

[1] The Conservation of Habitats and Species Regulations 2010 SI 490
[2] Council Directive of 30th November 2009 on the conservation of wild birds (2009/147/EC)
[3] Council Directive of 21/5/92 on the conservation of natural habitats and of wild fauna and flora (92/43/EEC)
[4] Convention on wetlands of international importance especially as waterfowl habitat, Ramsar, Iran 2/2/71 as amended by the Paris protocol 3/12/92 and the Regina amendments 3/6/87.
[5] Refer paragraph 48 Case C-127/02 *Waddenzee*

This report aims to provide an analysis of authoritative decisions which considered potential effects on land or sea outside a European site, but which provided supporting habitat, or other ecological function, linked to the designated area or its qualifying features, which can serve as a source of reference for advisers and decision makers.

A.4 The importance of case law to the decision making process

Case law is a vital source of information regarding how legislation should be correctly interpreted and applied. The Habitats Regulations transpose the requirements of the EU Wild Birds Directive and the EU Habitats Directive into domestic legislation. They set out a suite of legal obligations and responsibilities for a broad range of statutory agencies and decision making bodies (known as 'competent authorities'). As with all statutory instruments of this nature, there is scope for inconsistency in how the statutory provisions are interpreted and applied.

Too strict an interpretation might lead to plans or projects being delayed, subject to unnecessary restrictions, or ultimately refused under circumstances which were not intended to be incompatible with the underlying Directives. This can result in increased costs to, and frustration for, project proposers, which might have been avoidable, or unnecessary impediments to economic growth and development.

Too lenient an interpretation carries different risks. Plans or projects might go ahead without sufficient consideration of the potential harm to the sensitive habitats and species for which the sites have been designated. This in turn might lead to the deterioration of protected habitats and species, or a legal challenge through either the domestic or the European Courts regarding a failure to comply with the Regulations or the Directives.

Case law is therefore important in establishing a common understanding of how the tests involved in the assessment of plans and projects under the Habitats Regulations should be applied. There are credibility risks for decision makers, and those advising them, if a decision taken in respect of one proposed plan or project is not taken on the same basis as another plan or project, whether by the same or different competent authorities. Decision makers should strive to be consistent to ensure that the effects on the habitats and species protected under the Habitats Regulations are weighed appropriately and consistently in comparison with the benefits of proposals for change.

A.5 The meaning of 'authoritative decision'

Applying a 'plain English' interpretation, an 'authoritative decision' is a decision which has been subject to sufficient scrutiny, at an appropriate level, to impart a degree of *authority*.

In the context of this series of reports, 'authoritative decisions' are limited to those of the European and domestic (UK wide) court judgments and rulings (see A.6 and A.7 below), Secretary of State, or the Scottish or Welsh Ministers and certain Planning Inspector (in Scotland Reporter) decisions in respect of a proposed plan or project (see A.8 and A.9 respectively), and certain legally enforceable management measures such as a bye-law or statutory order (included in Secretary of State decisions in A.8). Article 6(4) 'opinions' from

the European Commission are also regarded as sufficiently authoritative to be included (A.10).

However, in this report all of the decisions referred to were made by the UK domestic courts, a Secretary of State, or an Inspector so, whilst the authority of decisions of the European Courts and 'Opinions' of the European Commission are introduced in A.6 and A.10 respectively, to provide relevant context, they are not considered further.
All these authoritative decisions are explained in the following sub sections so that they can be better understood in respect of:

a) how they should be read in relation to each other (some authoritative decisions carry greater weight than, or may supersede, other decisions); and
b) how they should be read in relation to a case which might currently be under consideration (where the reader is seeking guidance from this report as to a decision to be made).

A.6 Decisions of the European Courts

The relevant European court was the European Court of Justice until 1st December 2009, when the provisions of the Lisbon Treaty came into force and the court became known as the Court of Justice of the European Union. For the purpose of this report, all cases are referred to simply as those of the 'European Court'.

The European Court has two principal functions. Firstly, deciding cases of dispute between, on the one hand, the European Commission (EC), seeking to enforce the terms of the Directives; and, on the other hand, member states, who may be accused by the EC of failure to comply with the Directives. In these cases the European Court issues 'judgments' following consideration of written material and oral hearings. A judgment issued in the case of such a dispute is referred to in the documentation in terms of an 'action' of the court, because the decision reached by the court carries direct consequences for the parties involved.

The European Court also provides 'preliminary rulings'. These are not intended to resolve a dispute in the European court itself, but to answer questions submitted to the European Court by a court of a member state. Questions will almost invariably relate to how the domestic court of the member state should properly interpret the Directives when making a judgment in their own court. These decisions are also included in the term 'judgments'. The documentation relates to the 'reference' or 'request' made to the court rather than an 'action' related judgment in the case of a dispute.

This report uses the generic term 'judgment' in respect of European Court decisions, unless it is important to distinguish that a particular case was a 'ruling'. All judgments of the European Court carry the greatest weight because they are binding on member states in terms of both decision making and domestic court proceedings.

Importantly, all judgments of the European Court are accompanied by an 'opinion' from an Advocate General of the Court. The Advocate General's opinion is published in order to inform the Court's judgment. The relevant opinion exerts considerable influence over the

respective judgment. Opinions are also helpful because they often include more information concerning the details of the case concerned. The Advocate General's opinion carries less weight than the final judgment and the opinions are not binding on member states. However, they are so influential and carry such weight in European Court judgments and rulings that they are regarded as 'authoritative decisions' in the context of this research.

European Court decisions are binding on member states. They must therefore be given due weight by competent authorities and the courts of member states. They provide the definitive interpretation of how the Directives should be interpreted. However, not all areas of potential uncertainty have been the subject of a case in the European Court. In the absence of a judgment from the European Courts, the UK Courts may need to make decisions based upon their own interpretation.

A.7 Decisions of the UK Courts

Decisions taken in the UK Courts, which are of relevance to the application of the Habitats Regulations arise from judgments in the "High Court", the "Court of Appeal", and the "Supreme Court".

Relevant legal proceedings will start in the High Court, and if the High Court judgment is not referred to the Court of Appeal it will stand. However, if a High Court judgment is referred to the Court of Appeal the latter judgment will prevail and the legal principles established are binding on subsequent High Court judgments. Similarly, if a Court of Appeal judgment is referred to the Supreme Court the latter judgment will prevail and the legal principles established are binding on all lower courts including the Court of Appeal.

In Scotland, the Outer House of the Court of Session is equivalent to the High Court and the Inner House of the Court of Session is equivalent to the Court of Appeal.

A.8 Decisions of the Secretary of State / Scottish or Welsh Ministers

A decision taken by a Secretary of State, or an equivalent decision made by the Scottish or Welsh Ministers (the Ministers) is regarded as authoritative because it has been considered by a Government Department and signed off at a Ministerial level. It will usually (for example in the case of orders for development consent) be accompanied by or contain a detailed record of the related Habitats Regulations Assessment. Relevant decisions made by a Secretary of State or the Ministers relate to one of the following:

- an application for an 'Order for Development Consent' under the provisions of *The Planning Ac*t 2008 for a 'Nationally Significant Infrastructure Project'; or
- a consent required by a Secretary of State under primary legislation, for example, under the Electricity or Pipeline Acts; or
- in respect of a 'call-in' application, or a 'recovered' appeal under the provisions of the *Town and Country Planning Act* 1990 and related legislation (see further below), or
- the confirmation of a bye-law or other kind of statutory Order.

A decision made by a Secretary of State or the Ministers stands unless revoked or modified by them, or it is quashed by a Court because it has been challenged and found by the Court

to be unlawful. The grounds for such a challenge are limited and do not relate simply to the planning merits of the decision.

The Secretary of State and the Ministers also have powers to require a local planning authority to refer an application to them for their own determination, referred to as a 'call in' of a planning application. An Inspector (in Scotland a Reporter) will be appointed to conduct a local public inquiry and to report and make recommendations to the Secretary of State or the Ministers as the case may be. The Secretary of State and the Ministers follow established policies as to when they consider it to be appropriate to 'call-in' a planning application, but they are likely to do so if, for example, a local planning authority was minded to grant a planning permission that could have a significant adverse effect on a European site, against the advice of the statutory nature conservation body and in the face of national policy.

Where an applicant is aggrieved by a decision of a local planning authority to refuse permission for a development, or to grant it only subject to conditions that the applicant finds unacceptable, they have the right to appeal against the decision. The appeals are normally determined by a Planning Inspector or Reporter, (see A.9 below) but certain types of appeal can be 'recovered' for decision by the Secretary of State or the Ministers. Again the Inspector or Reporter will normally proceed to conduct a local public inquiry and report with recommendations to the Secretary of State or the Ministers. In both 'call-in' and 'recovered' cases the Secretary of State and the Ministers are not bound to accept the Inspector's or Reporter's recommendations.

A.9 Decisions of Planning Inspectors and Reporters

Planning Inspectors (and in Scotland planning Reporters) are the decision maker (the competent authority in the terms of the Habitats Regulations) in their own right in respect of all delegated appeals against the decisions of local planning authorities, which are not 'recovered' by the Secretary of State. Appeals are considered by way of an exchange of written representations (the majority of cases); or by way of an exchange of written material followed by a public 'hearing', or in a small proportion of cases, considered by a prior exchange of written material followed by the calling and examination of evidence at a local public inquiry, conducted by the Inspector making the decision. In the context of this report, the most authoritative decisions of Planning Inspectors / Reporters are regarded to be those which have followed a public inquiry, because in these cases the evidence has been subject to particularly intense scrutiny and the parties will have had the opportunity to make legal and other submissions to the Inspector or Reporter, however 'hearing' cases may also be regarded as sufficiently authoritative where evidence has been subject to particular scrutiny.

Planning Inspectors also conduct the 'examination' of local development plan documents submitted to the Secretary of State, in order to test them for 'soundness' before they can be adopted. The Inspector's report to the local planning authority is binding, but it is the authority who adopts the plan, having made any changes required by the Inspector's report.

A.10 Article 6(4) Opinions of the European Commission

Under the provisions of Article 6(4) of the Habitats Directive, it is open to a member state to seek an opinion from the European Commission (EC) as to whether the justification for authorising a particular plan or project would amount to 'imperative reasons of overriding public interest'. These are cases where the competent national authority cannot ascertain that there would not be an adverse effect on a European site, because a priority habitat or species may be adversely affected. This would normally rule out the consideration of economic or social reasons to authorise the project, but the option is available to seek an opinion as to the merits of the case from the EC. If the EC agree that the plan or project can proceed, they will examine compensatory measures and advise the member state accordingly. These are regarded as 'authoritative decisions' in the context of this research, because they have been scrutinised by the EC and the Commission's opinion is published. These opinions are also helpful because in making the case as fully as possible, the member state must set out the details of the effects of the project on the qualifying features and must explain in detail its proposed compensatory measures.

A.11 A note of caution

Given the large number of cases investigated, and the large volume of documents in relation to each case that had to be read, it was beyond the capacity of the researchers to undertake any investigations as to the accuracy of data, or to test the outputs of predictive models, or to undertake any other corroborative or verification work, as part of this research. All figures and factual information in this report are drawn directly from the documents which were read during the research. They are taken at face value. No assurance can therefore be given as to the accuracy or otherwise of information that was presented in the reports and decisions in the cases examined. For the purposes of this research it was sufficient to assume that all data recorded in the case reports and decisions were accurate and correct.

Having set out the basis on which this research considers a decision to be sufficiently 'authoritative' to be given weight in considering other decisions, it is worth bearing in mind that judgments stand unless superseded by a judgment in a higher court. Decisions made by the Secretary of State or an Inspector stand unless quashed by a Court, after having been challenged and found to be unlawful. Some decisions, and indeed, occasionally some domestic judgments, may not appear to be entirely consistent with established legal principles (for example those set by the European Court), or established approaches to decision making in terms of policy or scientific practice, but they nevertheless stand unless challenged or superseded. A judgment or a decision can only be made on the facts of the case as known at the time. If the evidence or arguments presented are incomplete or misleading the outcome may be affected. The application of case law evolves over time. Some judgments (or decisions taken in light of judgments at the time) may have been made before an important legal principle was established by a subsequent judgment.

For example, the Briels ruling in 2014 required a modification to the approach previously taken in respect of distinguishing mitigation and compensatory measures. Decisions made prior to this ruling did not have the benefit of that interpretation by the European court but were lawful and compliant at the time they were made.

Furthermore, no two cases are the same. What may initially appear to be inconsistency might, on closer examination, be a proper response to differences between the particulars of two cases which otherwise appear, at face value, to be equivalent.

It may be necessary, therefore, to consider the date of a decision or the extent to which a particular case is consistent with previous judgments or practice before relying upon it in a decision-making process.

It is the responsibility of the reader therefore to interpret and apply the findings in this report appropriately. The findings and conclusions of the report should be considered fairly, as a whole, and not quoted, used or applied selectively, in order to support a pre-determined or preferred conclusion.

B Why ‘functional links’ to European sites are important to decision making

B.1 What is meant by ‘functional linkage’

In the context of this report, the term ‘functional linkage’ refers to the role or ‘function’ that land or sea **beyond the boundary** of a European site might fulfil in terms of supporting the populations for which the site was designated or classified. Such an area of land or sea is therefore ‘linked’ to the site in question because it provides a (potentially important) role in maintaining or restoring a protected population at favourable conservation status. Whilst areas beyond a site boundary might serve a function in respect of a designated habitat type, for example by being linked hydrologically to the qualifying habitat, in the context of this report ‘functional linkage’ refers only to land or sea which is linked to a qualifying species (whether an Annex II species for which a SAC has been designated, or a bird species for which a SPA has been classified).

Whilst the boundary of a European site will usually be drawn to include key supporting habitat for a qualifying species, this cannot always be the case where the population for which a site is designated or classified is particularly mobile. Individuals of the population will not necessarily remain in the site all the time. Sometimes, the mobility of qualifying species is considerable and may extend so far from the key habitat that forms the SAC or SPA that it would be entirely impractical to attempt to designate or classify all of the land or sea that may conceivably be used by the species. Thus, for some sea birds the SPA may be confined to the cliffs where the sea birds breed, and will not extend to their feeding areas, which may be many kilometres away, or to the routes used by the birds either to reach their feeding grounds, or on migration, or during the winter. In respect of bats, for example, the SAC may need to be confined to the key roost sites used for hibernation, resting or breeding. The majority and, in many cases, the whole of the foraging areas of the bats in that roost, and the ‘commuting’ routes (flyways) between the foraging areas and the roosts, cannot practically be included in the SAC.

This approach to the definition of boundaries for SACs, where a qualifying species is mobile, is reflected in Article 4(1) of the Habitats Directive which states:

“...For animal species ranging over wide areas these sites shall correspond to the places within the natural range of such species which present the physical or biological factors essential to their life and reproduction. For aquatic species which range over wide areas, such sites will be proposed only where there is a clearly identifiable area representing the physical and biological factors essential for their life and reproduction...”

In practical terms, if the boundaries of a designated site were drawn to include *all* land or sea which might serve some function, at some point in time, in terms of the population for which the site had been designated or classified, the strict protection afforded by Article 6 would be applied more extensively than would be necessary to meet the objectives of the Directives. This could potentially place unnecessary restrictions on plans and projects which might not otherwise be required. By way of example, sites designated for harbour porpoise would need to potentially include vast areas of sea, if the boundaries were drawn to include all the areas

which might possibly provide some degree of support, at some point in time, for a given population. Regulatory procedures would be imposed on plans and projects simply on the basis that a harbour porpoise *might* occasionally feed or travel through the area affected by them.

The concept of 'functional linkage' is best explained in the context of the protection afforded to species under the Directives. EC guidance[6] on the strict protection of animal species of community interest refers to the Habitats Directive laying down two main 'pillars' which aim to meet the Directive's broad objectives. The first is linked to the protection afforded through designation of SACs (Articles 4-6 of the Directive) and the second is the protection of individuals of the 'European protected species' throughout a Member State's territory under Article 12. In summary:

"In order to achieve its objectives, the Habitats Directive provides for two main instruments: the Natura 2000 network of protected sites and the species protection provisions. The provisions for species protection apply to the whole of a Member State's territory and concern the physical protection of specimens as well as their breeding sites and resting places. Both regimes allow for exceptions under certain conditions. Both instruments are complementary and jointly aim to ensure a favourable conservation status for all species of Community interest."

The favourable conservation status of a given species is not therefore limited to how the species fares within the defined boundary of designated SACs. The guidance states at paragraph 17 that:

"Assessing and evaluating the conservation status of habitats and species within the Natura 2000 network is therefore not always enough, especially when the occurrences of habitats or species are only partly covered by the network, maybe even in some cases only to a relatively small extent."

Likewise, the Birds Directive refers not only to the protection of species within special protection areas, with reference to the network of SPAs, Article 4(4) provides:

"...Outside these protection areas, Member States shall also strive to avoid pollution or deterioration of habitats".

In practice, therefore, in respect of protected species, three potential situations arise as set out below. This report is relevant to the second of these three situations:

1. Land or sea within the boundary of a SAC or SPA which supports a qualifying species is strictly protected by the assessment approach set out in Article 6 (commonly referred to in the UK as a 'Habitats Regulations Assessment');
2. Supporting habitat in areas beyond the boundary of a SAC or SPA which are connected with or 'functionally linked' to the life and reproduction of a population for which a site has been designated or classified should be taken into account in a

[6] Guidance document on the strict protection of animal species of Community interest under the Habitats Directive 92/43/EEC, February 2007.

Habitats Regulations Assessment. However, that assessment will need to determine how critical the area may be to the population of the qualifying species and whether the area is necessary to maintain or restore the favourable conservation status of the species. Effects which would not be acceptable within the boundary of a European site may or may not be acceptable if they occur on functionally linked land or sea;

3. Individuals of a protected species, whether or not they are part of a population for which a European site has been designated or classified, are in any event afforded protection under the Directives, including in most cases their breeding and resting places, wherever they may be, inside or outside of the designated area. Species listed under Annex IV of the Habitats Directive are protected under the provisions of Article 12 as a 'European protected species'. Wild birds are similarly protected under the provisions of the Birds Directive.

The provisions of Article 6(3) and 6(4) of the Habitats Directive, as applied through a Habitats Regulations Assessment, do not apply to individuals of European protected species or individuals of species of birds which are not part of a population for which a European site has been designated or classified.

B.2 How functional linkage relates to the Habitats Regulations Assessment process

The concept of functional linkage is therefore an important consideration in decision making under the Habitats Regulations because the tests arising from Article 6(3) and 6(4) will need to be applied in respect of plans or projects which may significantly affect such supporting habitat and its contribution to the favourable conservation status of the relevant species.

Figure B.1 on the next page, provides an outline of the four stage process of Habitats Regulations Assessment. **The key question which this report is looking to address is how decision-makers have applied the Habitats Regulations Assessment (HRA) process where effects on species relate to functionally linked land or sea (situation 2 above) rather than the area within the boundaries of the relevant European site (situation 1 above)**. Few plans or projects will progress to stages 3 and 4 so the majority of the authoritative decisions referred to in this report concern the stage 1 'screening' test and the stage 2 'appropriate assessment' and 'integrity test'. These initial stages are briefly introduced below, but the relevance of functionally linked land (FLL) to the Habitats Regulations Assessment process is encapsulated in the following quote from paragraph 27 of the High Court judgment in RSPB and others v Secretary of State and London Ashford Airport Ltd [2014 EWHC 1523 Admin]:-

"There is no authority on the significance of the non-statutory status of the FLL. However, the fact that the FLL was not within a protected site does not mean that the effect which a deterioration in its quality or function could have on a protected site is to be ignored. The indirect effect was still protected. Although the question of its legal status was mooted, I am satisfied …. that while no particular legal status attaches to FLL, the fact that land is functionally linked to protected land means that the indirectly adverse effects on a protected site, produced by effects on FLL, are scrutinised in the same legal framework just as are the direct effects of acts carried out on the protected site itself. That is the only sensible and purposive approach where a species or effect is not confined by a line on a map or boundary

fence. This is particularly important where the boundaries of designated sites are drawn tightly as may be the UK practice".

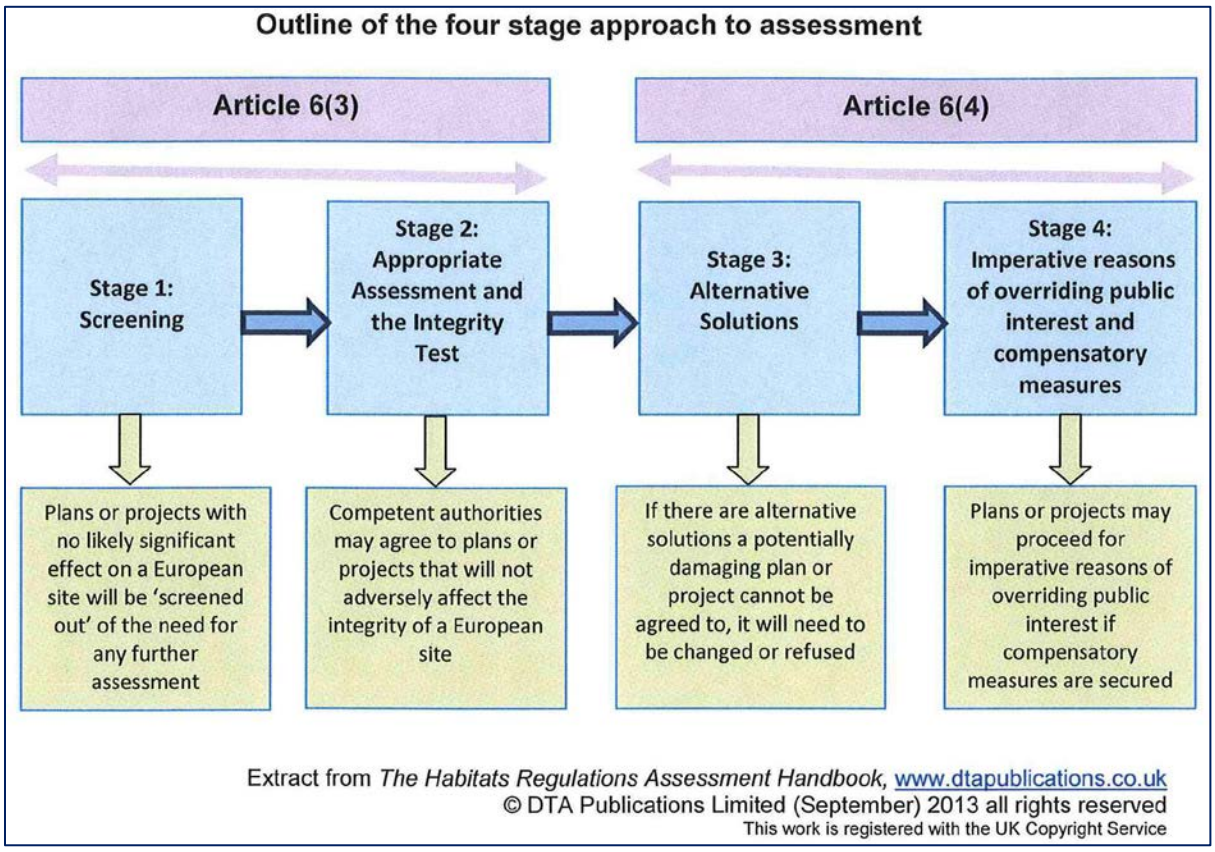


Figure B.1: Outline of the four stage approach to a Habitats Regulations Assessment

Stage 1: the 'screening' test

If it is not directly connected with or necessary to site management the decision-maker must determine whether a proposed plan or project is likely to have a significant effect[7] on the site. The decision on whether an appropriate assessment is necessary should be made on a precautionary basis. This is in line with the European Court's ruling in Case C-127/02 hereafter referred to as the Waddenzee judgment[8], which states that:

"*any plan or project not directly connected with or necessary to the management of the site is to be subject to an appropriate assessment of its implications for the site in view of the site's conservation objectives if it cannot be excluded, on the basis of objective information, that it will have a significant effect on that site, either individually or in combination with other plans or projects*."

Taking account of advice from the statutory nature conservation body, they should consider whether the effect of the proposal on the site, either individually or in combination with other

[7] Regulation 61(1)(a)
[8] *Landelijke Verenigning tot Behoud Van de Waddenzee, Nederlandse v Vereniging tot Bescherming von Vogels v Straatssecretaris Van Landbouw, Natuurbeheer en Visserij* (C-127/02: [2005] Env. LR14 [ECJ])

proposals[9], is likely to be significant in terms of the ecological objectives for which the site was designated, classified or listed. The statutory nature conservation body in England and its territorial waters out to 12 nautical miles (nm) is Natural England. Beyond that, in offshore waters, it is usually the Joint Nature Conservation Committee (JNCC), but arrangements have been made in some cases for Natural England to be the single consultee for both jurisdictions for projects, such as offshore wind farms, which may straddle the 12nm limit.

If a plan or project would not be likely to have a significant effect on the site alone, it should nevertheless be considered in combination with other plans and projects to establish whether there may be a significant effect arising from their combined impacts.

Stage 2: The 'appropriate assessment' and 'integrity test'

If the decision-maker concludes that a proposed plan or project not directly connected with or necessary for site management is likely to significantly affect a European site, they must make an 'appropriate assessment' of the implications of the proposal for the site in view of the site's conservation objectives[10]. These relate to each of the qualifying features for which the site was designated, classified or listed and will be provided by the statutory nature conservation body. The scope and content of an appropriate assessment will depend on the nature, location, duration, frequency, timing and scale of the proposed project and its effects, and the qualifying features of the relevant site. It is important that an appropriate assessment is made in respect of each qualifying feature for which a likely significant effect has been identified, and for each designation where a site is designated, classified or listed under more than one international obligation.

In the Waddenzee judgment, the European Court ruled that an appropriate assessment implies that all the aspects of a plan or project which can, by themselves or in combination with other plans or projects, affect the site's conservation objectives must be identified in the light of the best scientific knowledge in the field.

In the light of the conclusions of the appropriate assessment, the decision-maker must determine whether it can ascertain that the proposal will not adversely affect the integrity of the site(s)[11]. This test incorporates the precautionary principle. It is not for the decision-maker to show that the proposal would harm the site, in order to refuse the proposal. It is for the decision-maker to consider the likely and reasonably foreseeable effects and to ascertain that the proposal will not have an adverse effect on the integrity of the site before it may grant permission. If the proposal would adversely affect integrity, or the effects on integrity are uncertain but could be significant[12], the decision-maker should not grant permission, subject to the provisions of regulations 62 and 66, which relate to alternative solutions, imperative reasons of overriding public interest and compensatory measures. These are not discussed further in this report because they are not relevant to the research.

[9] Regulation 61(1)(a)
[10] Regulation 61(1)
[11] Regulation 61(5)
[12] See *ADT Auctions Ltd v Secretary of State Environment, Transport and the Regions and Hart District Council* (2000) JPL 1155 at p. 1171 where it was held to be implicit in the wording of reg 61(5) that the adverse effect on the integrity of the site had to be a significant adverse effect.

In the Waddenzee judgment, the European Court also ruled that a plan or project may be authorised only if a decision maker has made "*certain*" that the plan or project will not adversely affect the integrity of the site. "*That is the case where no reasonable scientific doubt remains as to the absence of such effects*." Decision-makers must be "*convinced*" that there will not be an adverse effect and where doubt remains as to the absence of adverse effects, the plan or project must not be authorised, subject to the derogation procedure outlined in Article 6(4) of the Habitats Directive[13].

The integrity of a site is the coherence of its ecological structure and function, across its whole area, which enables it to sustain the habitat, complex of habitats and/or the levels of populations of the species for which it was classified, designated or listed[14].

In determining the effect on site integrity, the advice of the statutory nature conservation body, the conservation objectives and any additional representations will need to be carefully considered. The UK courts have held that considerable weight should be given to the representations of the statutory nature conservation body and their advice should be adopted unless there are cogent and compelling reasons not to do so[15].

As part of the judgement on integrity, the decision-maker must consider the way in which it is proposed to carry out the project and whether conditions or other restrictions would enable it to ascertain that site integrity will not be adversely affected[16]. The decision-maker should consider whether a consent could be issued in accordance with regulation 61 subject to conditions. In practice, this means that it should identify the potential risks so far as they may be reasonably foreseeable in light of such information as can reasonably be obtained, and put in place a legally enforceable framework with a view to preventing the risks from materialising[17].

B.3 How functional linkage might influence the stage 1 and 2 conclusions

The concept of functional linkage is relevant to both the stage 1 screening decision and the stage 2 integrity test. In terms of the screening decision the extent to which an effect might 'undermine the conservation objectives' where it occurs beyond the boundary of the European site will be influenced by the role or function that the area serves and its importance to the maintenance of the population for which the site has been designated, classified or listed. An effect over a very small area of functionally linked land or sea, which is rarely used, might not undermine the conservation objectives, whilst the same effect over an area which provides prime feeding ground would be of more concern. A point will be reached where, in spite of an effect occurring while the species is beyond the boundary of a

[13] Regulation 62
[14] *Habitats Regulations Assessment* draft guidance from Defra July 2013, and formerly in Government Circular: *Biodiversity and Geological Conservation – Statutory Obligations and their Impact within the Planning System*. ODPM Circular 06/2005
[15] *R (Akester and Anor) v DEFRA and Wightlink Ferries* [2010] EWHC 232 (Admin)
[16] Regulation 61(6)
[17] See *WWF-UK Ltd and RSPB – v – Secretary of State for Scotland et al* [1999]1 C.M.L.R. 1021 [1999] Env. L.R. 632 opinion of Lord Nimmo-Smith

European site, the nature of the effect would be considered to undermine the conservation objectives for the qualifying species and an appropriate assessment would be required.

Turning to the stage 2 integrity test, in light of the accepted definition of integrity quoted in B.2 above, a site's integrity is inextricably linked to the concept of sustaining the population of a species for which the site has been designated, classified or listed. Where functionally linked land is necessary for that population to be so sustained then it must be *linked* to the site's integrity.

C The Case Studies

C.1 Selection

The researchers compiled a list of potentially relevant cases drawn from:

i. their own library of decisions, and their empirical knowledge of case work;
ii. a further web-based search of European Court judgments and opinions;
iii. a web-based search for decisions relating to nationally significant infrastructure projects and projects consented under the Electricity and Pipeline Acts in England and Wales and their territorial and UK offshore waters; and
iv. suggestions made by officers in Natural England following an e-mail enquiry of case officers by the research project manager.

The researchers were aware of many cases in the terrestrial environment across the UK, where decision-makers considering onshore wind farm proposals had taken account of land that may be functionally linked to SPAs or SACs, for example as a flight path or as a feeding area for birds or bats. Cases have been determined in all administrative jurisdictions, including decisions by Secretaries of State, Scottish Ministers, planning Inspectors and Reporters, under the provisions of the Town and Country Planning and Electricity Acts. The cases go back over 15 years, when the first examples arose, in Islay and the Mull of Kintyre, where wind farm proposals were refused by the then Secretary of State for Scotland, because of the potential collision risk of geese as they flew between their roost in an SPA and their feeding grounds in the countryside beyond. It was clearly impractical to attempt to identify, summarise and analyse all such onshore wind farm cases which involved consideration of functionally linked land.

Consequently, this report selects one of the most recent Electricity Act cases (E.22 Frodsham), a planning Inspector decision under the planning acts (E.25 Parkhead Farm) and a High Court case (E.7 Hargreaves) as representative of these many cases. These three adequately illustrate the issues and, along with the other cases in the report, help to answer the brief's particular emphasis in respect of the level of evidence required. These cases therefore represent many others in terms of a well-established principle that offsite habitat loss, or offsite displacement, disturbance or collision risk of SPA birds, triggers the application of the Habitats Regulations Assessment process. This is the case irrespective of the location of the project.

On the other hand, the recent range of issues arising in case work relating to the offsite effects of offshore wind farms, bear more detailed consideration. So all of the offshore wind farms which were nationally significant infrastructure projects, determined between 2013 and when the research was undertaken in January / February 2015, are considered in detail because they raise different issues and require different approaches to those now established in the onshore wind farm case work.

In light of the above, in the research for all three reports in January and February 2015, over 180 cases were identified as being relevant to the assessment of plans and projects affecting European sites. Following an initial screening exercise, twenty five of these

decisions were selected for detailed examination in this review, being relevant to the consideration of how decision-makers have taken account of functionally linked areas. One Court of Appeal case ***Boggis*** (E.9) is not directly relevant to functional linkage, as defined in section B.1, because the effects under consideration related to coastal processes and how they may affect an SPA, rather than effects on qualifying species whilst they were beyond the boundary of the SPA. It is included to illustrate the principles underlying evidence requirements so, except in respect of the discussion on evidence requirements, this case is excluded from further analysis in section D below.

Of the twenty four cases subject to further analysis, 19 referred to effects on bird species, 4 were related to effects on bats, and two referred to effects on Atlantic salmon. One of the 19 SPA cases also identified potential effects in respect of harbour porpoise and seals (E.10 Hornsea).

C.2 The summary table

Table C.1 below lists, in date order, the cases which were identified as relevant to this report, in that effects on functionally linked land or sea were material to the decision taken. Individual summaries of these cases are available in the Appendix to this report (Part E), with further analysis provided in part D 'Discussion and conclusions'.

Table C.1:

a) identifies the case by reference, title or familiar short title;
b) provides the date of the decision (or principal decision) and the decision-maker;
c) refers to the quality of the supporting evidence (see further discussion after the table);and
d) states the nature of the functional linkage.

The abbreviation 'SoS' in the table stands for Secretary of State.

Table C.1: Summary table of the 25 Cases reviewed in this report

Case	Date	Supporting Evidence	Nature of functional linkage
E.1: Abbotskerswell PC v Teignbridge DC	16/12/14 High Court	Good	Strategic flyways and sustenance zones for bats from South Hams SAC located in the plan area
E.2: Lydd Airport	16/05/14 High Court	Good	Supporting habitat for several SPA species adjacent to airport site.
E.3: Forest of Dean FoE v Forest of Dean DC (1)	01/05/14 High Court	Good	Strategic flyways and sustenance zones for bats
E.4: Buckinghamshire CC v SoS and HS2 Ltd	15/03/13 High Court	Good	Supporting habitat for over-wintering SPA wildfowl species
E.5: Forest of Dean FoE v Forest of Dean DC (2)	20/06/13 High Court	Good	Strategic flyways and sustenance zones for bats
E.6: Shadwell Estates	11/01/13 High Court	Poor beyond 1,500m buffer	The 1,500m buffer zone from the SPA and areas where at least five nesting attempts had occurred for stone curlew outside the SPA was sufficient, no evidence of a deterrent effect beyond it

Table C.1: Summary table of the 25 Cases reviewed in this report			
Case	**Date**	**Supporting Evidence**	**Nature of functional linkage**
E.7: Hargreaves v SoS	02/08/11 High Court	Good	Supporting habitat for over-wintering pink-footed geese
E.8: Sandale Developments	31/03/10 High Court	Good	Supporting in-river habitat for Atlantic salmon
E.9: Boggis	20/10/09 Court of Appeal	Poor	Asserted that sea defence scheme would affect coastal processes within SAC/SPA along coastline.
E.10: Hornsea Project One offshore wind farm	10/12/14 SoS	Based on assumptions about flight distances	Collision risk assessed for two SPAs (each more than 100km away). Also potential effects on harbour porpoise individuals which might be part of populations for which 26 pSCIs/SCIs designated (closest 44km, furthest 512km). Also on grey seal and harbour seal from two pSCIs at 44km and 64km respectively
E.11: Walney Extension offshore wind farm	07/11/14 SoS	As above	Collision risk assessed for 6 SPAs ranging from 20 - 287km from windfarm
E.12: Burbo Bank Extension offshore wind farm	26/09/14 SoS	As above	Collision risk assessed for 5 SPAs ranging from 6 - 55km from windfarm
E.13: North Killingholme power station project	11/09/14 SoS	Good	Supporting habitat for SPA species close to the SPA
E.14: Rampion offshore wind farm	16/07/14 SoS	Based on assumptions about flight distances	Collision risk assessed for SPA up to 490km distance
E.15: East Anglia 1 offshore wind farm	17/06/14 SoS	As above	Collision risk assessed for 3 SPAs; one at 54km, the other two at 259km
E.16: Able Marine energy park	28/08/13 SoS	Good	Function of important black-tailed godwit roost would be lost with loss of intertidal feeding areas
E.17: Triton Knoll offshore wind farm	11/07/13 SoS	Based on assumptions about flight distances	Collision risk assessed for SPA 83km away
E.18: Galloper offshore wind farm	24/05/13 SoS	As above	Collision risk assessed for SPA population from site 26.9km away.
E.19: Preesall Saltfield under-ground gas storage	09/04/13 SoS	Good	Supporting habitat used by pink-footed geese
E.20: Heysham to M6 Link Road	19/03/13 SoS	Good	Some birds known to feed and nest in affected fields but NE considered intensity of use was low and no particular fields favoured, with alternative sites nearby
E.21: Hinkley Point C nuclear power station	19/03/13 SoS	Good	Six large fields used for roosting by SPA populations
E.22: Frodsham onshore wind farm	19/10/12 SoS	Good	Site was a dredging disposal ground which hosted notably high numbers of over-wintering and passage waders from SPA immediately adjacent
E.23: Portsmouth Stadium	15/06/95 SoS	Good	Prime winter feeding ground for Brent geese from nearby SPA
E.24: Lemonford Caravan Park	06/03/14 Inspector	Poor	Strategic flyway for greater horseshoe bats

Table C.1: Summary table of the 25 Cases reviewed in this report			
Case	**Date**	**Supporting Evidence**	**Nature of functional linkage**
E.25: Parkhead Farm	11/05/09 Inspector	Good	Collision risk to geese from SPA located 5km away

The assessment of the quality of the supporting evidence is a professional judgment by the researchers based on the evidence available. It relates to how well it was demonstrated that there was, or probably was, a functional linkage between the area of land or sea affected and the population of the species for which a site had been classified or designated. Where this is assigned as 'good' it indicates actual survey or other evidence and relatively clear (or even obvious) links to the SAC/SPA.

As described in section D below, other cases, in the marine environment, had to be based on reasonable scientific assumptions. This is because assumptions had to be made about where the birds in the development area might breed and about recorded foraging distances. Nevertheless, these cases should not be regarded as having a weak evidence base for the links. In the precautionary approach of the Habitats Regulations sufficient evidence pointed to a possibility or a risk of an effect on SPA populations.

In the case of E.20, there was good evidence that SPA bird species used the affected area for nesting and feeding, but there was agreement that the use of the area was so low that the proposed link road would not have a significant effect. There was a functional link, but not an important one, emphasising that the two decisions are different: Is there a functional link? Is there a significant effect on the European site?

In respect of three cases, the evidence base was considered to be 'poor' with links not well established. The first of these was Boggis, already discussed above. The second was in Breckland (E.6) where the court found there was no convincing evidence that birds would be deterred from breeding by housing located beyond the 1,500m buffer zone agreed in the adopted local plan. In the other case (E.24) the developer declined to carry out surveys that could have demonstrated the functional link that the Inspector and the planning authority clearly considered to be likely.

D Discussion and conclusions

D.1 Cases concerning effects on bats

Four cases reviewed (E.1, E.3, E.5 and E.24) concerned effects on bat SACs. An analysis of the cases confirms the following points.

The qualifying features of the bat SACs were the population of the bats using the SAC as a roost. Each individual of the relevant species is protected in its own right as a European protected species, so too are its breeding and resting places, whether it or the places are in the SAC or anywhere outside it. However, there is additional protection for the animals through designation of their key roost areas as SACs. Furthermore, if the bats in the SAC rely on habitat outside the SAC, the loss or reduction (in area or ecological value) of which could have a significant effect on the population of the SAC bats, then that supporting habitat should be treated in a Habitats Regulations Assessment in the same way as the habitat in the SAC. In short, whilst the European protected species legislation protects the bats and their breeding and resting places, it is the effect of the designation of the SAC that protects, through the Habitats Regulations Assessment process, the habitat of the bats outside the SAC.

The issues in the four cases examined related to the potential loss, interruption, or diminution of the ecological value of the routes (flyways) used by the bats from the SAC to reach their foraging grounds, which were widely spread around the countryside beyond the SAC boundary. Hence the bats would be indirectly affected by way of loss of habitat, or by the interruption or severance of the flyways or by the introduction of deterrent effects in the flyways and/or in the foraging areas. Reduction in ecological value of the foraging areas and/or impediments to the bats reaching their foraging areas could undermine achievement of the conservation objectives of the SACs and therefore affect the conservation status of the bats in the SACs.

In all cases the risk to the population of bats, for which the SAC had been designated, arising out of effects which could occur beyond the boundary of the SAC was accepted. In the case of Forest of Dean FoE v Forest of Dean DC (2) (E.5) the judgment stated: *"Although the Northern United compound has not been designated a SAC, the designation of an area as a SAC protects the population of the species using the site and not just the site itself".*

In the Lemonford case (E.24) the Inspector noted:

"the South Hams SAC is self-evidently an important area in biodiversity terms and its functionality in terms of the strategic flyways is clearly fundamental to its integrity as habitat"

In three of the four cases the approach to functional linkage was underpinned by substantive survey work showing a reasonable likelihood that the area that would be affected was functionally linked to the relevant SAC bat population. But in the case of the Lemonford appeal (E.24), it was the uncertainty that arose from the inadequacy of the survey work that led to the dismissal of the appeal. The appellant had been reluctant to undertake the survey work recommended by the planning authority's ecological adviser and Natural England. If

carried out, this survey could have established the relative importance of the area of offsite habitat that would have been affected and the likely efficacy of mitigation measures.

The Inspector recognised that regulation 61 of the Habitats Regulations must be applied in respect of the potential changes that the development would make to the ecological function of the flyway for the bats from the SAC. The information requested by the planning authority in this case was not regarded as onerous. The applicant / appellant had declined to submit it so the Inspector concluded: "*Appropriate assessment cannot, in my view, be adequately undertaken on the basis of the information to hand".* On the precautionary basis of the regulations he dismissed the appeal for this and other reasons.

D.2 Cases concerning effects on bird species

Twelve of the 19 cases involving SPAs related to terrestrial or coastal habitats involving a range of waterfowl species, mainly geese, ducks and waders. Seven related to the marine environment and sea birds.

Terrestrial and coastal SPAs

In all twelve cases involving terrestrial or coastal SPAs the decision maker recognised the potential importance of functionally linked land and that it should be treated as part of the Habitats Regulations Assessment. Table D.1 below summarises the 12 cases. It indicates the case reference number in section E, the type of development proposed, the distance of the development from the SPA, the key issue and whether the decision concluded no likely significant effect (LSE) or an adverse effect on integrity (AEOI) of the SPA.

Table D.1: Cases involving effects on birds associated with terrestrial or coastal SPAs

Development	Distance	Issue	LSE / AEOI
E.2 Airport expansion	<1km	Disturbance and displacement of several species of feeding and roosting birds on fields located varying distances from the SPA by aircraft noise and operational activity	No LSE
E.4 High speed rail	8km	Disturbance and displacement by construction and operational noise of railway c.400m from a water body used by wildfowl from an SPA 8km away. Subject to further research LSE unlikely	No LSE
E.6 Residential	1.5km buffer	Disturbance and displacement of breeding stone curlew as a result of residential development near to nesting sites. Plan included a buffer ensuring no new residential development within 1.5km of the SPA boundary or areas outside SPA of suitable habitat and where at least 5 nesting attempts had been recorded since 1995	No LSE beyond buffer
E.7 Wind turbines (2)	5km	Pink-footed geese roosted in the SPA and fed on fields adjacent to the development 5km from the SPA. Without mitigation collision mortality risk was up to 50 geese per year. Mitigation measures accepted as sufficient to avoid a significant effect	No LSE

Table D.1: Cases involving effects on birds associated with terrestrial or coastal SPAs			
Development	**Distance**	**Issue**	**LSE / AEOI**
E.13 Gas power station	Adjacent	Large numbers of golden plover, curlew, lapwing and other species used fields close to the development site. Construction noise could disturb them and wildfowl on a nearby wetland if birds were in the zone of influence of 500m. Noise mitigation measures accepted as sufficient to avoid a significant effect	No LSE Beyond buffer zone
E.16 Marine energy industry	Adjacent	Large numbers of black-tailed godwit could be displaced because of loss and reduced functionality of intertidal areas in the SPA, compounded because they relied on adjacent wetland pits outside the SPA for roosting so would be displaced from both areas	AEOI
E.19 Underground gas storage	Adjacent	Loss of small area of land used by feeding pink-footed geese from SPA owing to construction of access road. Displacement effect likely to be within 500m of construction activity, on edge of large area used by geese	No LSE
E.20 Road link	3km	Proposed road would pass through fields used for breeding and feeding by birds from SPA. NE advised intensity of use was light, species did not favour any particular field and alternative areas also outside the SPA were available to them	No LSE
E.21 Nuclear power station	Adjacent	Fields to be used during construction supported significant numbers of waterfowl from 2 adjacent SPAs requiring substantial mitigation measures during construction and operation of a lay-down area	LSE but no AEOI due to conditions
E.22 On shore wind farm	Adjacent	Site was a dredging disposal ground which hosted notably high numbers of over-wintering and passage waders, redesign and additional mitigation measures secured	No LSE
E.23 Football stadium	<1km	Development site considered to be prime winter feeding grounds for dark-bellied brent geese from nearby SPA, refused permission, effectively concluding AEOI could not be ruled out	AEOI
E.25 Wind turbines	5km	Potential for feeding habitat loss and significant collision risk for pink-footed geese flying to other feeding grounds from SPA roost. Mitigation package agreed with NE and RSPB	No LSE

In each of the above twelve cases there was some degree of evidence that birds from the relevant specific SPA (or reasonably assumed to be birds from that SPA) were actually using, or were highly likely to use, the area affected by the development. In the case of the stone curlews in Breckland, the issue was securing a buffer zone that was large enough to ensure that breeding birds would not be disturbed by proposed development, the calculation of the buffer zone being based on scientific survey and analysis including historical records of breeding attempts.

In the terrestrial or coastal environments, three points became clear from the research:

a) the scope for SPA bird species to use land, whether in close proximity or further away from the SPA, is often limited by urban development, land use patterns, noisy or other disturbing activities or operations, barriers and of course suitability of the habitat, even where there is open land or water;
b) there are often good quality, pre-existing records, such as Wetland Bird Census data, to indicate the use of specifically defined areas outside the SPA by birds;

c) surveys of use, or potential use, of land or water bodies by relevant SPA species is usually reasonably obtainable even if surveys are required over a period of time.

Consequently, in the terrestrial and coastal environments, the possibility of the presence of functionally linked land is more readily identifiable, and the land areas more easily defined as relatively discrete areas, than in the marine environment discussed below. Where pre-existing records were not available, but an area affected by a development appeared likely to be used as functionally linked land, new survey work was undertaken to establish the level of use in order to inform a Habitats Regulations Assessment, screening or appropriate assessment, as the case may have been. Even where good data about levels of use were available, survey effort was continued to improve understanding, case E.22 Frodsham onshore wind farm, being a case in point.

Sea bird SPAs

In contrast, the seven sea bird SPA cases relating to offshore wind farms had to approach the potential of functionally linked areas of sea differently. Pre-existing survey information that was location specific to the development sites was scarce. All new proposals involved surveys to establish the use of the proposed wind farm areas by all species of birds. It was then necessary to consider whether the birds that were recorded in these offshore areas may reasonably be assumed to be individuals associated with an SPA.

In some cases individual breeding birds such as gulls, were tagged in the SPA and tracked to see if they ventured into the development area. In other cases, however, such tracking studies would have been impractical. So the likelihood of an area being functionally linked to an SPA was established by considering the likelihood of the birds having come from an SPA. This depended in part on the proportion of that species which bred in SPAs, and the distance that the area lay from SPAs for the relevant species, compared to the known recorded foraging distance or migratory behaviour of the species. For example, if all pairs of a species which was recorded feeding in, or otherwise using, a wind farm area, bred in an SPA, and one or more SPA for that species was located within the known maximum recorded foraging distance from the wind farm site, it could reasonably be assumed that the birds that could be affected in the breeding season were linked to one or more such SPA; and the wind farm area was functionally linked.

Whilst in respect of kittiwakes some assessments considered their seasonal migration routes, the critical distance was usually the known maximum breeding season recorded foraging distance, or in some cases the known breeding season flight paths, which varied considerably from one species to another. No standard cut off distance from an SPA could be used as a surrogate for the risk of a significant effect. All of this was relevant before the ornithological analyses attempted to calculate collision risk or displacement for the birds that were recorded as using the development area. In Habitats Regulations Assessment terms it was first necessary to establish the likelihood of the birds potentially affected being qualifying features of an SPA somewhere. Inevitably this involved some assumptions.

Some of the distances involved, between a particular wind farm and an SPA potentially significantly affected, may appear to be considerable, and were sometimes considered by applicants to be over-precautionary. However, on the scientific evidence in the Habitats

Regulations Assessments accompanying the Secretary of State's decision in each of these cases, the calculations of displacement and collision risk modelling relate to bird populations which can reasonably be assumed to be those relating to an SPA. In all cases the SPA(s) potentially affected was specifically identified.

Relevant to this discussion, Table D.2 below indicates the proportion of the British population of the relevant species of sea bird which breed in a UK SPA.

Table D.2: Relative percentage of GB breeding population of seabird species which are associated with the UK SPA network	
Seabird species	**% of GB population breeding in UK SPA suite**
Gannet *Morus bassanus*	98%
Kittiwake *Rissa tridactyla*	78%
Manx Shearwater *Puffinus puffinus*	100%
Lesser black-backed gull *Larus fuscus*	100%
Sandwich tern *Sterna sandvicensis*	72%

In the case of effects upon gannet, manx shearwater and lesser black-backed gull therefore, where site surveys identify individuals within the project area, it is reasonable to assume that they are from a population for which an SPA has been classified. The uncertainty may sometimes relate to which SPA they were associated with. In the case of sandwich tern, where 72% of the breeding population are within the SPA network, it might be considered reasonable (on a precautionary basis) to assume that at least a proportion of individuals within a project area would be likely to be associated with an SPA. However, in the case of the Triton Knoll offshore wind farm (E.17) the development was thought to be located at the upper limit of the tern's foraging range and, although sandwich terns were recorded in the area, the prospect of the SPA terns reaching the area was considered to be low.

Table D.3 attempts to encapsulate the evidence about sea areas considered to be functionally linked in these seven cases. It indicates, for each of the main sea bird species, their maximum recorded foraging distance, and then the case reference number in section E, the name of the development and the distance of the development from the SPA, with an indication as to whether the decision-maker concluded a likely significant effect (LSE) or an adverse effect on integrity (AEOI) of the SPA.

Table D.3 Maximum foraging distances of sea birds and location of offshore wind farm case studies		
Species	**Maximum foraging**	**Cases / distance from SPA(s)**
Gannet *Morus bassanus*	c.590km	E.14 Rampion 490km LSE in combination no AEOI
		E.15 East Anglia One 259km LSE in combination no AEOI
		E.17 Triton Knoll 83km LSE in combination no AEOI
Kittiwake* *Rissa tridactyla*	c.60km / c.120km	E.14 Rampion 490km no LSE alone or in combination
		E.15 East Anglia One 259km no LSE alone or in combination
		E.17 Triton Knoll 83km no LSE alone or in combination
Manx Shearwater *Puffinus puffinus*	330km	E.11 Walney Extension 120km, 147km, 287km LSE in combination no AEOI

Table D.3 Maximum foraging distances of sea birds and location of offshore wind farm case studies		
Species	**Maximum foraging**	**Cases / distance from SPA(s)**
Lesser black-backed gull *Larus fuscus*	c.140km	E.12 Burbo Bank 6km, 6km, 42km, 55km LSE no AEOI
		E.15 East Anglia One 54km LSE no AEOI
		E.18 Galloper 27km (surveys tracked one bird 159km from the SPA)
Sandwich tern *Sterna sandvicensis*	Uncertain	E.17 Triton Knoll 47km thought to be upper limit of likely range, species recorded in the area no LSE

*as mentioned in the text, possible effects on kittiwakes during seasonal migration were considered but also concluded not to be a likely significant effect.

D.3 Cases concerning effects on Atlantic salmon

The Sandale case (E.8) concerned effects on individuals of Atlantic salmon, when they would be upstream of the SAC boundary. With the project being located upstream of the designated SAC, the functional linkage between the population for which the SAC had been designated and the individuals potentially affected by the proposed development is clear, because all salmon in the upper reaches must be salmon who had migrated through the SAC.

In the case of Sandale, the *risk* to the SAC from effects on individuals while they were within the Camowen River was considered to be sufficient to require further investigation. These risks had not been adequately assessed when the original permission had been granted. The evidence regarding the presence of Atlantic salmon was from a desktop study supported by 'reports' of individuals being observed.

In the case of Burbo Bank the location of the project site meant that the noise from piling activity had the potential to prevent Atlantic salmon from making their seaward migration from the SAC out into the Irish Sea. The risk to the smolt migration could, however, be avoided through a proposed timing restriction on the driving of the piles. This enabled a conclusion of no adverse effect on the integrity of the site. Whilst the SAC was located 32km away from the project area, the mouth of the estuary through which the individuals had to migrate to reach the Irish Sea was only 8km from the project area.

D.4 Cases concerning harbour porpoise and seals

Only one case subject to review, Hornsea (E.10), considered functional linkage in the context of marine mammals. The effects considered related only to displacement, rather than displacement and mortality as in the case of the sea birds. Hornsea identified potential effects on the populations of harbour porpoise for which 26 pSCIs/SCIs mostly in the territorial waters of other member states, had been proposed or designated and the populations of grey seals and harbour (common) seals from which two pSCIs had been proposed by the Netherlands.

Whilst on-site surveys revealed that individuals of the species were present within the project area, the links to populations for which a SAC had been designated were necessarily based on assumptions. The distance from the European sites, and the range over which the

species were known to forage for food being so extensive, particularly in the case of harbour porpoise, meant that risks from displacement were not considered to represent a threat to the integrity of any of the sites affected, for any of the species for the reasons indicated in the following quotations (emphasis added).

"it is unlikely that harbour porpoise from these European sites are solely using the Hornsea area to feed in. This is particularly unlikely given the highly mobile and wide foraging nature of harbour porpoises and their ability to feed on a range of prey sources.

"Given the relatively small maximum displacement rate (1.7 km from piling work) predicted for harbour seals relative to their typical foraging range (40-50 km), it is unlikely that even if all the offshore wind farms in the North Sea are undertaking concurrent or sequential piling activity that harbour seals from Doggersbank pSCI, and the Klaverbank pSCI will be without sufficient foraging habitat given their diverse diet preferences and foraging range.

D.5 Evidence requirements

Extending the discussion on evidence requirements in D.1 and D.2, as to the initial identification of an area that may be functionally linked, this section examines the evidence requirements for the assessment of the effects on the species.

The case of Boggis (E.9) established some important principles with regard to the evidence in respect of a perceived *risk* to a European site (emphasis added).

"a claimant who alleges that there was a risk which should have been considered by the authorising authority so that it could decide whether that risk could be "excluded on the basis of objective information", must produce credible evidence that there was a real, rather than a hypothetical, risk which should have been considered".

The level of supporting evidence relating to risks concerning functional linkage varies across the cases reviewed. In the cases of terrestrial plans and projects affecting SPA bird species, survey information was available and the use of affected areas by the populations for which the SPA had been classified was plainly established. Of relevance to the outcome of the decisions was the level of use of the functionally linked land affected and its importance to the maintenance of the populations for which the site had been classified.

In the case of the North Killingholme Power Project (E.13), whilst the applicant's Environmental Statement found that the area adjacent to the project was important for SPA bird species, with findings of up to 10,000 golden plover on land to the north of the development representing 2.5% of the British wintering population (and exceeding the threshold for SPA classification in its own right), the maximum area over which the birds were likely to be affected by the proposed visual and noise disturbance effects was small (within 500m). With reference to this 'zone of influence' *"The SoS is satisfied that there would be no adverse effect on SPA/ Ramsar birds outside the designated site boundary from construction disturbance as only a small population of birds are recorded in close proximity and disturbance effects will be mitigated by using hoarding and barriers to screen operations."*

Likewise, in the case of the Heysham to the M6 link road (E.20), although birds from the SPA were known to utilise the fields alongside the proposed route, Natural England did not consider functional linkage effects merited appropriate assessment on the basis that "*the intensity of use by these species was light and species do not favour any particular field with suitable alternatives available nearby*".

In contrast, in the case of Hinkley power station (E.21), surveys revealed that more than 1% of the Severn Estuary SPA populations and approximately 25% of the Somerset Levels and Moors SPA population of golden plover were affected while on functionally linked land. The land was clearly important to the maintenance of the SPA populations and (emphasis added):

"The Secretary of State has considered the potential disturbance and displacement effects on birds that are features of the Severn Estuary SPA and Ramsar. In view of the numbers of birds close to the foreshore and the sensitivity of those species, most notably Shelduck, he considers it necessary to impose a range of mitigation measures during construction and operation to reduce disturbance due to noise, artificial light, vessel movements and the presence of personnel and machinery on site.

" ...The Secretary of State has considered the potential impacts on bird species and the assemblage of birds that are the feature of the Somerset Levels and Moors SPA/Ramsar. For the same reasons that apply to the Severn Estuary SPA/Ramsar (covered in chapter 7 of this document), he concludes that, with the relevant DCO and EA Environmental Permit requirements in place, the impacts from HPC alone and in combination would not have an adverse effect on site integrity."

Case E.22, Frodsham onshore wind farm, is also a notable example of the high level of confidence in the data obtained about the use of functionally linked land. Although the immediate proximity of the site to the SPA and the locally well-known ornithological value of the application site would always have indicated a high probability of a functional link, this case is characterised by a considerable survey effort and extensive analysis in order to fully understand the use of the site by birds and its links to the SPA. Furthermore, the proposal was redesigned and resubmitted in a variation to the application, as a result of this detailed work assessing the ecological functionality of the site. As a result of this effort and redesign, together with the mitigation package, Natural England withdrew its objection and the Secretary of State concluded that, despite the high levels of use of the site by SPA populations and the immediate proximity to the SPA, there would be no likely significant effect on the SPA or the Ramsar site either alone or in combination with other plans or projects.

The adequacy of the assessment undertaken in respect of functional linkage is also influential. A perceived threat or risk from effects occurring beyond the boundary of the site was identified in the cases of Shadwell (E.6), Sandale developments (E.8) and Lemonford caravan park (E.23). The first case concerned birds, the second Atlantic salmon and the third bats. But all cases considered a 'risk' related to functional linkage and the extent to which that risk had been adequately addressed.

In the case of Shadwell, the approach adopted by the Council had been to create a buffer zone of 1,500m from the SPA boundary and likely offsite breeding areas for the stone curlew. In court it was argued that an allocation beyond 1,500m and in an area where the birds had been observed to breed in the past ought not to have been included in the plan. But the court gave considerable weight to the extensive survey work, analysis and expert opinions expressed by Natural England and the RSPB. The challenge centred on how the underlying reports had been interpreted and applied and further risk (beyond that already addressed in the agreed approach) based on anecdotal reports. With reference to the risk identified in this challenge, the court ruled that "*The evidence... is nowhere near providing the 'cogent and compelling' reasons that are needed in order to depart from the views of a statutory consultee.... No new evidence has been produced which undermines the validity of the 1,500 metre distance".* And then making a reference to Boggis (E.9) "*In order to succeed on ground 3, Shadwell has to produce credible evidence of a real risk to the integrity of the SPA".* In other words the court was satisfied that the competent authority, in consultation with the statutory nature conservation body, had correctly identified the risks from effects on functionally linked land and had undertaken the necessary assessment work and developed an agreed approach based on the evidence obtained. The risk raised by the claimants lacked evidence of equivalent credibility.

In both Sandale Developments and Lemonford Caravan Park a credible risk was identified to the qualifying features of the European site from effects on functionally linked areas but, in both cases, there was insufficient evidence that these risks had been adequately assessed. A precautionary approach was thereby taken. In the absence of evidence upon which the credible risks could be excluded by obtaining relevant information and making an assessment of the significance of the effects on the qualifying features (salmon and bats), the Sandale decision was found not to satisfy the requirements of the Regulations and the Lemonford appeal was dismissed for want of adequate information to make an appropriate assessment.

"the underlying lack of specific information about the manner in which the site is actually used by the Greater Horseshoe Bat militates against the robustness of conclusion that is in this instance required."

"Appropriate assessment cannot, in my view, be adequately undertaken on the basis of the information to hand." (Lemonford (emphasis added))

"In light of the applicant's evidence about protected salmon above the watercourse in the Camowen River the potential impact is clear and must be excluded"

"More particularly, the information that has now emerged does indicate that the risk exists that the proposed development will have significant effects on the SAC, in that the salmon may be affected by discharges into the watercourse. The precautionary principle dictates that the risk exists because it cannot be excluded on the present state of knowledge" (Sandale (emphasis added)).

In essence, the analyses in D.1 to D.5 show that, amongst other things:

a) The identification of an area as functionally linked land in the terrestrial or coastal environment is generally relatively straightforward and readily recognised, but may sometimes not be apparent and may require some initial survey and analysis or collation of pre-existing data, to establish the link;
b) The identification of an area as functionally linked sea is more challenging and has to be approached differently for marine developments; nevertheless an approach in respect of sea birds and marine mammals appears to be developing and although necessarily relying to a greater extent on assumptions, it provides a robust approach which is suitably precautionary without being onerous;
c) Once identified as functionally linked land or sea, the evidence required by decision makers in stages 1 and 2 of the Habitats Regulations Assessment process are no different to those that might reasonably be expected in relation to direct or on-site effects on the European site. The precautionary principle applies equally to functionally linked land and sea. Where effects might be significant and there is insufficient information to ascertain that there would not be an adverse effect on the integrity of a site, in terms of the population of the species for which the site has been classified or designated, authorisation has been denied; consistently with the provisions of the Regulations. However in the majority of cases sufficient evidence was available for the decision-maker to conclude that there would be no significant effect, or no adverse effect on site integrity, if the project was authorised.

Technical Guidance Series

CIEEM Chartered Institute of Ecology and Environmental Management

In association with amphibian and reptile conservation

Competencies for Species Survey: Natterjack Toad



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1. Introduction

The purpose of this document is to set out the knowledge, skills and experience required to survey, disturb or to carry out research works (hereinafter referred to generically as 'survey') for the natterjack toad, *Epidalea calamita*, in a professional capacity. To be undertaking such work, CIEEM would consider attainment of the criteria outlined in this guidance to be a minimum for an individual to competently survey for natterjack toads. The Institute aims to drive up standards in the ecological profession for the benefit not only of ecologists but also of the public.

This document does not provide guidance on the techniques to be employed in carrying out survey activities: references to published sources of guidance on survey methods are listed in Section 5.

2. Status

The natterjack toad has a localised distribution in the UK, and has undergone substantial historical declines due to habitat loss and fragmentation. The remaining strongholds are dune systems and upper saltmarshes along the coasts of Merseyside, Cumbria and Dumfriesshire; with smaller and more isolated coastal populations in Lincolnshire, Norfolk, Suffolk, Kent, Dorset, Denbighshire, Flintshire, Cheshire and Lancashire. There are a few lowland heathland populations in Hampshire and Surrey. There are also coastal populations in County Kerry, southwest Ireland. Natterjacks are absent from Northern Ireland. The few inland populations in southeast England represent a great reduction from their historical range, largely due to loss of heathland. In coastal areas, their range has shrunk due to urbanisation and loss of habitat due to other land-use changes.

The natterjack toad is a 'pioneer-opportunist' which relies upon sparsely-vegetated habitats with shallow, ephemeral ponds. There are around sixty populations in the UK, with only about 3,000 breeding females in total (ARC 2009). The natterjack toad is a UK BAP Priority Species, with a long-standing reintroduction programme, and intensive conservation efforts. Inbreeding depression has contributed to the decline of some isolated populations, and chytrid fungus has been cited as another possible cause of declines. Although declining in the north of its European range, the natterjack toad is listed as a species of least concern on the IUCN Red List because it is locally abundant across much of its range.

3. Legislation and Licensing

Legislation

Natterjack toads are protected under the following legislation. This outline of legislation is not comprehensive and the appropriate legislation should always be consulted for a definitive list of offences.

COUNTRY	LEGISLATION GIVING PROTECTION	SCHEDULE OR ANNEX LISTING
England	Conservation of Habitats and Species Regulations 2010 Wildlife and Countryside Act 1981 (as amended)	Schedule 2 Schedules 5
Wales	Conservation of Habitats and Species Regulations 2010 Wildlife and Countryside Act 1981 (as amended)	Schedule 2 Schedules 5
Scotland	Conservation (Natural Habitats, &c.) Regulations 1994 (as amended)	Schedule 2
Republic of Ireland	European Communities (Natural Habitats) Regulations 1997 (as amended) Wildlife Act 1976 (as amended)	First schedule Schedule 5

Licensing

The following Statutory Nature Conservation Organisations (SNCOs) are responsible for issuing licences to permit actions that would otherwise be illegal under the relevant legislation:

COUNTRY	SNCO
England	Natural England
Wales	Natural Resources Wales
Scotland	Scottish Natural Heritage
Republic of Ireland	National Parks & Wildlife Service

A *survey licence is required:* for any survey work using techniques likely to disturb natterjack toads at any site where there is a reasonable likelihood that they are present. *e.g.* terrestrial searches or netting.

A *survey licence is not required*: for non-intrusive surveying such as listening for the natterjack toad's distinctive call, or for spawn string counts.

In the event that a natterjack toad is encountered *e.g.* when conducting a spawn string count, an unlicensed surveyor must cease survey work, unless they are in the presence of a surveyor with an appropriate licence.

It is the role of the appropriate licensing authority to define the criteria for issuing such licences as may be required for survey work, therefore, attainment of the skills set out is no guarantee that a licence will be obtained; other criteria will need to be considered.

4. Knowledge, Skills and Experience

To independently and competently undertake survey of natterjack toads an individual would be expected to possess all of the following knowledge, skills and experience.

A surveyor needs to recognise their level of attainment along a continuum. Those without the breadth and depth of the knowledge, skills and experience that CIEEM consider a minimum, should always work with, or seek advice from, an adequately experienced individual.

Knowledge

Individuals should have a knowledge and understanding of:

a. conservation status;
b. distribution;
c. threats to populations, species range and species survival;
d. natterjack toad ecology, breeding biology and behaviour;
e. known ecological requirements (*e.g.* ponds and terrestrial habitats);
f. legal protection;
g. licensing and permissions;
h. appropriate survey seasons;
i. current relevant guidance on survey methods and standards;
j. survey methods used to survey for natterjack toads (*e.g.* catching and handling and indirect methods such as spawn counts, tadpole and metamorph estimation, terrestrial searches, call identification and torchlight survey) and the strengths, weaknesses and limitations of these methods;
k. range of factors that might lead to bias in the survey results, and false negatives;
l. factors affecting surveying (*e.g.* temperature/weather conditions);
m. biosecurity procedures and disease transmission risks, in particular chytrid fungus;
n. sources of information on known occurrence and distribution of natterjack toads (including NBN Gateway, local biological/environmental records, county atlases and local contacts/Amphibian and Reptile Group(s));
o. metadata standards / data sharing; and
p. health and safety issues associated with surveying for natterjack toads.

Skills

Individuals should have skills and experience enabling them to:

a. confidently identify natterjack toads in all life stages, including adult males' breeding calls;
b. confidently distinguish natterjack toads from other native and non-native amphibian species that may be present;
c. assess habitat suitability of a site for natterjack toads;
d. scope fieldwork appropriately;
e. plan and implement sound scientific surveys;
f. select suitable temperature and weather conditions for survey;
g. carry out visual surveys;
h. make spawn or animal counts and classify population size classes;
i. interpret and analyse survey data (recognising any constraints on survey results);
j. take appropriate biosecurity precautions; and
k. take appropriate health and safety precautions.

If surveys involve netting or using refugia, individuals should also be able to:

a. survey ponds for natterjack toads (all appropriate life stages) using a net;
b. survey terrestrial habitat for natterjack toads using refugia;
b. legally and humanely handle natterjack toads; and
c. effectively record biometric data from live toads.

Practical Experience

Whilst some of the knowledge and skills set out in this guidance can effectively be gained from personal study, vocational training and structured education, applicants for natterjack toad survey licences would be expected to have gained practical experience (over a number of survey seasons) under the direct supervision of an experienced and licensed natterjack toad surveyor. Individuals planning to apply for their own natterjack survey licence should have the following specific experience:

a. performing surveys of natterjack toads across all life stages (spawn to adult), including capture and biometric measurement, in compliance with the *Natterjack Toad Conservation Handbook* (Beebee & Denton 1996); and

b. survey of at least 10 different breeding ponds, ideally at several different sites, under supervision of a natterjack toad licence-holder; and survey over at least five different occasions, spanning the breeding season April to July.

This experience should be fully recorded and documented in a portfolio which can be presented to the statutory authorities when a licence is applied for.

5. Reading

The following references are essential reading for those wishing to gain the necessary knowledge, skills and experience to survey for natterjack toads.

Beebee T and Denton J (1992) An evaluation of survey methods for studying natterjack toads (*Bufo calamita*) outside the breeding season. *Amphibia-Reptilia* **13**: 365-374.

Beebee T and Denton J (1996) *Natterjack Toad Conservation Handbook*. English Nature, Peterborough.

HCT (2009) *Natterjack Toad Species Action Plan*. The Herpetological Conservation Trust., Bournemouth.

The Herpetological Conservation Trust (undated) Natterjack Toad Survey Guidelines. The Herpetological Conservation Trust, Bournemouth.

Natural England (2011) *Standing Advice Species Sheet: Natterjack toads*. Natural England, Sheffield.

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CIEEM
Chartered Institute of Ecology and Environmental Management

GUIDELINES FOR ECOLOGICAL IMPACT ASSESSMENT IN THE UK AND IRELAND

Terrestrial, Freshwater, Coastal and Marine

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SUMMARY

The Chartered Institute of Ecology and Environmental Management (CIEEM) has developed these Guidelines to promote good practice in Ecological Impact Assessment (EcIA) relating to terrestrial, freshwater, coastal and marine environments in the UK and Ireland.

EcIA is a process of identifying, quantifying and evaluating potential effects of development-related or other proposed actions on habitats, species and ecosystems[2]. The findings of an assessment can help competent authorities understand ecological issues when determining applications for consent. EcIA can be used for the appraisal of projects of any scale including the ecological component of Environmental Impact Assessment (EIA). When undertaken as part of an EIA, EcIA is subject to the relevant EIA Regulations. Unlike EIA, EcIA on its own is not a statutory requirement. It is an evaluation process undertaken to support a range of assessments.

EcIA is a process that is most effective if all contributing ecologists and other specialists work in collaboration. An EcIA report (or the ecological chapter of an EIA Environmental Statement) should clearly and simply describe the significant effects of any project so that all interested parties understand the implications of what is proposed. These Guidelines explain the key elements of the EcIA process:

Chapter 1 – Introduction. Overview of the EcIA process and underpinning principles.

Chapter 2 – Scoping. Determining the matters to be addressed in the EcIA, including consultation to ensure the most effective input to defining the scope. Scoping is an ongoing process – the scope of the EcIA may be modified following further ecological survey/research and during impact assessment.

Chapter 3 – Establishing the baseline. Collecting information and describing the ecological conditions in the absence of the proposed project, to inform the assessment of impacts.

Chapter 4 – Important ecological features. Identifying important ecological features (habitats, species and ecosystems, including ecosystem function and processes) that may be affected, with reference to a geographical context in which they are considered important.

Chapter 5 – Impact assessment. An assessment of whether important ecological features will be subject to impacts and characterisation of these impacts and their effects[3]. Assessment of the significance of the residual ecological effects of the project (those remaining after mitigation), including cumulative effects.

Chapter 6 – Mitigation, compensation and enhancement. Incorporating measures to avoid, reduce and compensate negative ecological impacts and their effects, and the provision of ecological enhancements. Monitoring impacts and their effects. Evaluation of the success of proposed mitigation, compensation and enhancement measures.

Chapter 7 – Implications for decision making. Consideration of the legal and policy framework throughout the EcIA process.

1 INTRODUCTION

1.1 The purpose of these Guidelines is to promote good practice in Ecological Impact Assessment (EcIA) relating to terrestrial, freshwater, coastal and marine environments in the UK and Ireland.

1.2 The context of EcIA is well established. The EU Biodiversity Strategy[4] and national biodiversity strategies[5] reflect the need to conserve biodiversity[6] in the face of pressure from development, other land use change and climate change. These strategies reflect the aspirations of the Convention on Biological Diversity. International work on the benefits of nature to society (natural capital and ecosystem services), the UK National Ecosystem Assessment[7] and *The Economic and Social Aspects of Biodiversity in Ireland*[8], have increased understanding of the importance of conserving biodiversity for human wellbeing and the economy. This is reflected in national policy objectives for biodiversity and nature conservation, such as the Natural Environment White Paper[9] in England, the White Paper on Natural Resources[10] in Wales, the *Land Use Strategy for Scotland*[11], and the *National Biodiversity Action Plan 2017-2021*[12] in Ireland. EcIA supports implementation of national biodiversity strategies and national planning policies for safeguarding biodiversity and supporting the delivery of sustainable development. In the marine environment this is reflected through UK adoption of the Marine Strategy Framework Directive, a programme of measures designed to achieve or maintain Good Environmental Status in UK seas by 2020[13], the UK Marine Policy Statement[14] and the Integrated Marine Plan for Ireland (2012)[15].

1.3 *EcIA is a process of identifying, quantifying and evaluating the potential effects of development-related or other proposed actions on habitats, species and ecosystems.* EcIA can be used for the appraisal of projects of any scale: it is a systematic and repeatable process applicable to a wide range of projects.

1.4 Two particular uses of EcIA are:

- providing the ecological component of Environmental Impact Assessment (EIA) required under EIA Regulations
- demonstrating how a project accords with relevant planning policy and legislation where an EIA is not required.

1.5 The EcIA process can be fitted seamlessly into the EIA process or it can stand alone. Summary information on EIA is provided in Box 1. It is recommended that the EcIA process is used to present an assessment of the ecological effects of any project which could affect biodiversity.

Box 1: Environmental Impact Assessment

EIAs are carried out to meet the requirements of Council Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment, as amended by Council Directives 97/11/EC, 2003/35/EC and 2009/31/EC and redrafted in a codified version Directive 2011/92/EU. The amended Environmental Impact Assessment (EIA) Directive 2014/52/EU entered into force in 2014 to simplify the rules for assessing the potential effects of projects on the environment and Member States had to apply these rules from May 2017[16].

The Directive requires EIA to be carried out before development consent is granted for projects that are likely to have significant environmental effects. EIA is a mandatory requirement for projects listed in Annex I of the Directive. It is also required for projects that meet the criteria for Annex II development and are likely to have significant environmental effects (as defined in Annex III of the Directive). EIAs should provide a comprehensive understanding of the implications of a project proposal, including consideration of impacts on biodiversity, and interactions with soil, water and air.

The main statutory instruments that implement the Directive in the UK (updated in 2017) are available on the website for UK legislation[17]. In the Republic of Ireland, the EIA Directive was transposed in planning legislation in 2000 and given further effect in 2013[18].

1.6 Work undertaken for EcIA can inform Habitats Regulations Assessments (HRA), the Appropriate Assessment process in Ireland and Water Framework Directive Assessments. These Guidelines do not cover these Assessments, but a summary is provided in Box 2.

Box 2: Habitats Regulations Assessment, Appropriate Assessment in Ireland and Water Framework Directive Assessment

Projects affecting designated sites protected by the Habitats Directive[19] (transposed into national Regulations[20]) will require specific assessments in accordance with the Directive. Similarly, projects affecting water bodies may require assessment under the Water Framework Directive[21]. These Guidelines do not explain the specific assessment processes to be applied or the criteria governing decisions. Guidance on implementing the Habitats Directive is provided by the European Commission in *Managing Natura 2000 sites the provisions of Article 6 of the 'Habitats' Directive 92/43/EEC*[22] and in more detail in The Habitats Regulations Assessment Handbook[23] and in Appropriate Assessment of Plans and *Projects in Ireland: Guidance for Planning Authoritie*[24]. Information on the Water Framework Directive is available on the Joint Nature Conservation Committee website[25] and on the Irish Water Framework Directive website[26].

Information assembled for an EcIA that is relevant to subsequent Habitats Regulations Assessment (HRA) and/or Water Framework Directive assessments should be presented in a format that can be readily extracted for these other assessments.

Use of the Guidelines

1.7 These Guidelines should be followed by ecologists undertaking EcIA. The Guidelines also provide regulators, decision-makers and those submitting projects with an indication of the information needed to adequately consider projects in the light of biodiversity legislation and policy.

1.8 These Guidelines are applicable to ecologists acting for:

- a project proposer such as a developer submitting a planning application
- a competent authority making the decision about consent for a proposed project
- public bodies with biodiversity and landscape duties and/or Water Framework Directive duties / Marine Strategy Framework Directive duties
- consultees such as those who advise the competent authority in a statutory or voluntary capacity.

Differing Scales of EcIA and a Proportionate Approach

1.9 EcIA can be applied to projects of widely varying scales. The EcIA principles and process outlined in this guidance are relevant to all developments that may impact on ecological features – this term is used throughout to cover habitats, species and ecosystems. However, the level of detail required in an EcIA will inevitably be proportionate to the scale of the development and complexity of its potential impacts. These Guidelines do not prescribe exactly how to undertake an EcIA, but provide guidance to practitioners for refining their own methodologies.

1.10 Scoping (Chapter 2) should be proportionate to potential effects on ecological features. Professional ecologists need to use their knowledge and experience to judge the resources required to complete an adequate and effective EcIA.

1.11 Emphasis in EcIA is on 'significant effects' (see Box 3) rather than all ecological effects. Relevant legislation, regulations, plans and policies should be considered at an early stage, as these will have a bearing on the scope of investigations, how potential effects are interpreted, and the criteria needed for determining significance.

Box 3: Significant effects

Significance is a concept related to the weight that should be attached to effects when decisions are made. For the purpose of EcIA, 'significant effect' is an effect that either supports or undermines biodiversity conservation objectives for 'important ecological features' (explained in Chapter 4) or for biodiversity in general. Conservation objectives may be specific (e.g. for a designated site) or broad (e.g. national/local nature conservation policy) or more wide-ranging (enhancement of biodiversity). Effects can be considered significant at a wide range of scales from international to local.

A significant effect is an effect that is sufficiently important to require assessment and reporting so that the decision maker is adequately informed of the environmental consequences of permitting a project.

In broad terms, significant effects encompass impacts on structure and function of defined sites, habitats or ecosystems and the conservation status of habitats and species (including extent, abundance and distribution).

See paragraphs 5.24 – 5.40 for further explanation of significant effects.

A Rigorous and Transparent Approach

1.12 A scientifically rigorous and transparent approach to EcIA is essential. EcIA should be undertaken by qualified professionals with an appropriate level of experience in ecological survey and impact assessment who are recognised by a relevant professional body such as CIEEM.

1.13 All interested parties should understand the assessment process and who is responsible for implementing and monitoring the actions needed to deliver biodiversity objectives. The EcIA must provide reliable and defensible information about, and interpretation of, the likely significant ecological effects from inception to operation, maintenance and, where appropriate, closure and decommissioning.

1.14 Account needs to be taken of existing information and understanding, in conjunction with the results of EcIAs undertaken elsewhere. CIEEM encourages all practitioners to share data and results, for example through Local Record Centres (LRCs) in Britain[27], the Centre for Environmental Data and Recording (CEDaR) in Northern Ireland[28], the National Biodiversity Data Centre in Ireland[29] and the National Biodiversity Network Atlas Scotland[30]. Consideration should also be given to the publication of findings to allow review and learning by others, for example the Marine Data Exchange.[31] Guidelines for accessing and using biodiversity data, including the submission of data, have been produced by CIEEM (2016)[32].

1.15 The EcIA should consider the significant ecological effects of a project in the light of relevant planning policies and legislation. Legislation relating to terrestrial, freshwater, coastal and marine environments is evolving and practitioners should always check for changes and revisions. These Guidelines are based on requirements at the time of publication and changes in legislation and policy may necessitate periodic review. The Guidelines do not attempt to explain legislative detail, and users should refer to relevant legislation and case law or seek specialist legal advice.

1.16 EcIA benefits from early consultation with the relevant competent authority, Statutory Nature Conservation Bodies (SNCBs), Environment(al) Protection Agencies (EPA) and Non-Governmental Organisations (NGOs). Through EcIA the developer/project proposer can be made fully aware of matters such as site designations, protected habitats and protected species, and their implications, before pursuing a project. Engagement with consultees on a regular basis will help refinement of the proposal, smooth progression through the planning process and minimise misunderstanding and controversy.

1.17 There will be examples on the coast where a project will be subject to both marine and terrestrial consent processes. Early consultation provides an opportunity to agree the scope of the assessment and approach for the assessment process in these circumstances. In England a Coastal Concordat has been published which promotes co-ordination between regulatory authorities responsible for the range of consents, permissions and licences, and the provision of a single point of entry into the regulatory system for applicants[33].

1.18 Good communication is essential between ecologists and other professionals engaged in the assessment process (e.g. geomorphologists, hydrologists, social practitioners, EIA coordinators), together with proponents, to help inform judgements and refine proposals. It may also require engagement with comparable international bodies where trans-boundary effects are predicted e.g. in the marine environment.

Key Principles

1.19 The following principles[34] underpin EcIA:

Avoidance	Seek options that avoid harm to ecological features (for example, by locating on an alternative site).
Mitigation	Negative effects should be avoided or minimised through mitigation measures, either through the design of the project or subsequent measures that can be guaranteed – for example, through a condition or planning obligation.
Compensation	Where there are significant residual negative ecological effects despite the mitigation proposed, these should be offset by appropriate compensatory measures.
Enhancement	Seek to provide net benefits for biodiversity over and above requirements for avoidance, mitigation or compensation.

The EcIA Process

1.20 The EcIA process is summarised in Box 4. EcIA is an iterative process; for example, scoping is an ongoing process - the initial proposed scope of the EcIA may be modified following further ecological survey/research and during impact assessment.

Box 4: Summary of the EcIA process

Initial project design

At the outset of the project, the proponent's ecologist(s) should:

- obtain information on the project, any alternatives that have been studied and existing ecological information
- undertake a gap analysis of known and needed information and plan and prioritise gap filling
- review ecological implications of alternatives
- discuss key ecological considerations about the project design (and alternatives) with the project proposer and the design team (e.g. engineers, architects)
- recommend modifications to the design to avoid negative ecological impacts or, where this is not possible, to reduce them
- explore opportunities for ecological enhancements and net gain of biodiversity as early as possible

Screening (EIA only)

The project proposer may seek a formal screening opinion from the competent authority to determine the need for EIA under the EIA Regulations.

Scoping EcIA

Scoping determines the issues to be covered by the EcIA. Where an EIA is required, the competent authority must provide a 'scoping opinion' if requested by the developer. In all other cases, it is advisable to seek the views of competent authorities and key consultees on the proposed scope of the EcIA.
In all cases, scoping will need to:

- identify any potential licensing requirements for survey and/or development regarding legally protected species
- identify all proposed construction, operating, maintenance, closure and decommissioning activities that may generate significant ecological effects
- identify significant other developments that may give rise to cumulative effects
- collect initial baseline information, such as through a Preliminary Ecological Appraisal (PEA), to enable identification of potentially important ecological features within the zone of influence of the project
- identify for assessment those important ecological features that could sustain significant positive or negative impacts, as far as possible given the information available (and employing the precautionary principle)
- identify relationships with other environmental aspects e.g. water, landscapes
- identify data gaps
- propose suitable spatial and temporal scopes for the assessment and identify the main ecological issues to be addressed
- undertake preliminary assessment of potential ecological impacts on identified features, incorporating existing data/information
- reconsider spatial and temporal scope and amend the extent of preliminary investigations if necessary
- list those features that do not need further assessment, with appropriate justification
- identify suitable survey/research methodologies that (ideally) have been agreed with consultees
- confirm potential opportunities for avoidance of impacts, mitigation, ecological enhancement and biodiversity net gain
- ensure biodiversity data collected is suitable for potential use in biodiversity metrics for assessment of 'net gain' of biodiversity
- ensure compliance with standards and consistency with formal methods of evaluation.

For EIA projects, and as good practice for non-EIA projects, scoping should also:

- involve appropriate consultation with the competent authorities, statutory agencies, other regulatory bodies, and possibly relevant NGOs and other non-statutory consultees, regarding the project and the proposed scope of the assessment
- produce a scoping report that can be circulated for comment and modified accordingly.

Impact assessment

The EcIA team will be involved in the following assessment process, which should cover all phases of any project:

- determine the importance of ecological features affected, through survey and/or research and with reference to available contextual information
- assess impacts potentially affecting important features
- characterise the impacts, e.g. extent, magnitude, duration, reversibility, timing and frequency
- identify cumulative impacts
- identify significant effects of impacts in the absence of any mitigation.

The surveys and research that are undertaken may indicate that the scope of the assessment should be adjusted and further studies carried out.

Evolution of project design and mitigation

- consider alternative location(s) or layouts for the proposed development
- identify mitigation measures and explain their likely success
- identify opportunities for ecological enhancement and net gain of biodiversity

- design and agree monitoring strategy and monitoring of mitigation performance with the competent authority (and, in some cases, consultees)
- provide sufficient information for mitigation measures to be implemented effectively, e.g. through an Environmental Management Plan (EMP).

Identify significant residual effects and their legal, policy and development management consequences

- produce a clear summary of the residual impacts and the significance of their effects following incorporation of avoidance and mitigation measures
- consider the implications of significant effects on the important ecological features in accordance with planning policies and legislation
- where significant negative effects cannot be avoided or mitigated, identify compensation measures to be implemented
- include mitigation, compensation and enhancement measures in the EMP or similar
- make an assessment of the significance of any cumulative effects.

Reporting

The final EcIA report (or for EIAs, the Environmental (Impact) Statement) should clearly set out all the ecological information necessary for a robust decision to be made. Key aspects include a description of the following:

- ecological baseline and trends if the project were not to go ahead, including the survey data used to inform the baseline
- criteria used to evaluate ecological features
- criteria used to assess the significance of effects arising from the impacts of the project
- justification of methods used
- the identification of likely impacts (positive and negative) on ecological features together with an explanation of the significance of the overall effects for each important ecological feature
- mitigation, compensation and enhancement measures
- legal and policy consequences
- identification of any limitations to the assessment, or the surveys which underpin it, and an explanation of the implications
- a presentation of any analytical techniques used and the analysis itself.

The report should also set out the ecological monitoring required to:

- confirm the implementation of conditions/planning agreements
- audit predicted impacts against the actual situation
- take measures to rectify unexpected negative impacts and ineffective mitigation, compensation and enhancement measures.

EcIA reports

1.21 EcIA reports should be tailored to suit individual circumstances and different formats are acceptable. However, CIEEM considers it important that the structure and content of EcIA reports are standardised. A logical structure for a report is provided in *CIEEM Guidelines for Ecological Report Writing*[35] and is reproduced in Appendix 3. Where an EIA is required, the Ecological Impact Assessment will be presented in a way that fits the overall style and structure of the Environmental (Impact) Statement. However, the content of Appendix 3 remains relevant. Where elements of this content lie outside the presentation of the main Ecological Impact Assessment (usually an ecological chapter of the EIA), cross-reference should be included.

Note: The following definitions are used for the terms 'impact' and 'effect':

Impact – Actions resulting in changes to an ecological feature. For example, the construction activities of a development removing a hedgerow.

Effect – Outcome to an ecological feature from an impact. For example, the effects on a dormouse population from loss of a hedgerow.

6 AVOIDANCE, MITIGATION, COMPENSATION AND ENHANCEMENT

Introduction

6.1 A sequential process should be adopted to avoid, mitigate and compensate negative ecological impacts and effects. This is often referred to as the 'mitigation hierarchy' (see 1.19 and 5.1). For most projects, avoidance, mitigation, compensation and enhancement measures should be identified as part of the EcIA process.

Avoidance and Mitigation

6.2 Negative impacts should always be avoided where possible, for example by deciding not to locate a project in a particular area or making a change to scheme layout to ensure no negative impacts. Avoidance can also be part of mitigation. Mitigation includes measures to avoid or reduce the negative impacts of a project, for example careful timing of an activity to prevent an impact occurring.

6.3 Avoiding and/or minimising negative impacts is best achieved through consideration of potential impacts of a project from the earliest stages of scheme design and throughout its development. Many impacts can be avoided or reduced by consideration of alternatives.

6.4 Mitigation measures incorporated into the scheme design are often described as 'embedded mitigation' or 'mitigation by design'. This can include the re-design of the layout of the scheme, or adjusting the location of certain activities. Mitigation by design is particularly beneficial as there is greater certainty that it will be delivered. Wherever possible mitigation should be by design rather than left to a request by the competent authority. In Ireland, NPWS has produced guidance for developers in the construction and extractive sectors[112]. The efficacy and any potential knock-on effects of both embedded and non-embedded mitigation should always be assessed.

Compensation

6.5 Compensation describes measures taken to offset residual effects resulting in the loss of, or permanent damage to, ecological features despite mitigation. For example, it may take the form of replacement habitat or improvements to existing habitats. Compensation can be provided either within or outside the project site (defined by the red line of a planning application). Compensation should always be seen as a last resort, when all other mitigation options have been exhausted. Compensation is particularly difficult to achieve in the marine environment (see paragraph 5.36).

6.6 As a general rule, compensation should be focused on the same type of ecological features as those affected and equivalent levels of ecological 'functionality' sought. There will be cases when it is not possible to achieve ecological equivalence through compensation. Any replacement area should be similar in terms of ecological features and ecological functions that have been lost or damaged, or with appropriate management have the ability to reproduce the functions and conditions of those ecological features. Compensation should be provided as close as possible to the location where effects have occurred and benefit the same habitats and species as those affected.

6.7 Replacement ratios of compensatory habitat greater than one-to-one are frequently appropriate because of the uncertainty inherent in compensation, particularly in cases which require ecological restoration, habitat creation or translocation of species or habitats. The scientific basis for deriving appropriate ratios is not exact and will vary depending on the habitat or species concerned. Increased replacement ratios can also help take account of the time lag in delivering compensation and regaining the same maturity, complexity and diversity of habitats and the full complement of associated species.

6.8 Biodiversity offsets is a concept that provides a measurable approach to compensation with the aim of achieving 'no net loss'[113]. Box 19 summarises biodiversity offsets.

Box 19: No net loss and biodiversity offsets

Biodiversity offsets are a form of compensation which may be considered when a development is expected to have significant residual impacts on biodiversity despite planned mitigation measures. Biodiversity offsets have a formal requirement for measurable outcomes. The main requirement is to quantify losses (through effects, after avoidance and mitigation) and gains (through offsets) using the same 'metric', for example hectares of habitat of a particular quality[114]. Using an offset metric in this way provides for transparency of outcome and enables explicit demonstration of 'no net loss'. Such metrics can also be used to support the achievement of net gains (see paragraphs 6.11 to 6.13).

6.9 The delivery of compensation measures, including biodiversity offsets, is likely to involve access to land, or land purchase, outside a scheme footprint and a commitment to long-term management through legal agreements. They therefore require early consideration in project design. The principles of offsetting should be agreed with the competent authority at an early stage, particularly where this is not clearly set out in a policy or biodiversity offsetting strategy.

6.10 The distinction between mitigation and compensation can be difficult to determine. Where ecological equivalence can be delivered within the project site this is sometimes incorrectly considered mitigation rather than compensation. However, the correct distinction between mitigation and compensation is that mitigation avoids or reduces the occurrence of negative impacts and effects and compensation addresses effects which are residual, after avoidance and mitigation have been considered. Measures to address impacts and effects that will occur should therefore be referred to as compensation whether the compensation is located within or outside of the project site.

Enhancement

6.11 It is important that development is sustainable and, where possible, projects produce a net gain for biodiversity and nature conservation. National policies promote the inclusion of measures to enhance biodiversity within development proposals. Enhancement of biodiversity should be an objective of all projects. Principles for achieving net gain within development have been published by CIEEM, CIRIA and IEMA (2016)[115].

6.12 Enhancement is improved management of ecological features or provision of new ecological features, resulting in a net benefit to biodiversity, which is unrelated to a negative impact or is 'over and above' that required to mitigate/compensate for an impact. For example, mitigation for bats may involve erecting bat boxes in a woodland to replace suitable bat *roosting* features that have been removed, and the woodland habitat itself may be enhanced for *foraging* bats by increased woodland planting and the creation of glades. Enhancement could be linked to the delivery of wider socio-economic benefits such as wetland restoration and flood risk management. Enhancement measures should be described in the EcIA.

6.13 Enhancement measures should be designed to deliver biodiversity objectives that are specified in relevant policy documents, and evidence should be provided to support the likelihood of delivering the predicted benefit. They should be incorporated into scheme design and assessed within the EcIA. To ensure that enhancements are enduring, their delivery and management should normally be guaranteed through a legal obligation, such as, in England and Wales, a planning obligation under section 106 of the Town and Country Planning Act 1990[116], or its equivalent provision elsewhere.

Designing Mitigation and Compensation

6.14 The design of mitigation and compensation measures is an iterative process. It needs to consider what is realistically achievable and the likely extent to which success can be guaranteed, based on good practice guidance and evidence. Measures need to be agreed and a commitment made by the developer before a planning application is submitted to the competent authority. Such commitments must then also be agreed with the competent authority and secured through appropriate planning conditions, consents, permits and/or legal agreements. The aim should be to provide the competent authority with as much certainty as possible over the likely effectiveness and deliverability of the proposed measures (see BS 42020 clauses 6.6 and 8.1[117]).

6.15 Mitigation and compensation measures should address specific effects. For example, where a development will result in an effect on the conservation status of a population of great crested newts (or, for example, common frog and smooth newts in Ireland) through the loss of terrestrial foraging habitat, compensation should involve the provision of new foraging habitat rather than new breeding ponds. Mitigation and compensation measures should achieve long-term results. Their duration should match the duration of the impact.

6.16 Mitigation, compensation and enhancement measures should have defined criteria for success, which allows success or failure to be measured by monitoring. It is not appropriate to propose measures that cannot be delivered within a reasonable time frame, or simply present a wish-list of measures.

6.17 However, mitigation measures can include 'contingency measures' where there is uncertainty as to whether or not an effect will occur. For example, this may be relevant to effects that only occur if activities take place at particular times of year (such as site clearance during the bird nesting season), or as a result of unpredictable changes (such as where a badger sett, absent at the time of the EcIA, has been established prior to construction). In such cases the contingency measures should identify how such issues will be resolved and consider the possible effects on the project programme.

6.18 For many species, particularly those with legal protection, there is published guidance that describes appropriate approaches to mitigation. In some cases, it will be necessary to design new approaches to mitigate an effect, and the advice of relevant experts and statutory and non-statutory consultees should be sought. If standard methods are not being used, this will need to be explained and justified.

6.19 In some cases compensation measures will need to be in place and 'functioning' before the significant harm occurs. This is particularly likely where the effect is on designated sites or legally protected species. Lead-in times to deliver the measures will need to be carefully considered and explicitly stated in the EcIA.

6.20 The design of mitigation and compensation measures should be revisited with other members of the design team as there may be constraining factors or knock-on effects that need to be considered by other specialists: an integrated approach will increase successful delivery of the design. Should mitigation for one habitat or species have a negative effect on another, it may still be appropriate to implement the mitigation, although the negative effects will need to be considered in the assessment.

Delivery

6.21 Mitigation and compensation measures often carry a degree of uncertainty. Uncertainty associated with a design will vary according to a number of factors, and where necessary these should be discussed in the relevant section of the EcIA. Factors to be considered include:

- technical feasibility of what is proposed – using experience from projects where a similar measure has been carried out
- overall quantity of what is proposed – is it large enough to be viable? Is it of equivalent function to any habitat lost?
- overall quality of what is proposed – does it compare favourably with features lost or damaged?
- level of commitment – is there a realistic understanding of the resources and effort required to achieve predicted outcomes?
- provision of long-term management
- timescale for predicted benefits.

Reporting Mitigation, Compensation and Enhancement

6.22 The description of mitigation, compensation and enhancement measures within the EcIA must be sufficient to allow the competent authority and relevant stakeholders to see clearly how effects will be addressed. The level of detail needed will vary between schemes, between different measures within a scheme and should include quantity, location, timing, techniques and resources.

6.23 It is helpful to set out how a project has evolved in response to ecological considerations and to indicate how mitigation that has been incorporated into the scheme design has avoided or minimised negative impacts. Presenting the results of the assessment 'with' and 'without' mitigation allows the need

for mitigation and/or compensation to be clearly identified. Where mitigation is fully integrated into the scheme and there is high confidence that it will be implemented, it may be appropriate simply to assess the significance of effects of the mitigated project, with this assessment reflecting the likelihood of the incorporated measures being successful. Where there is any uncertainty, then the with/without mitigation approach to assessment described above should be used to ensure transparency.

6.24 Ideally, details of mitigation, compensation and enhancement measures will be incorporated into an Ecological Design Strategy (EDS) or Environmental Management Plan (EMP) which sets out methods and responsibilities for delivery. The plan should detail timescales for delivery and key criteria for judging success.

Monitoring

6.25 The EcIA should set out the ecological monitoring required to audit predicted impacts and effects against the actual situation. This will enable any necessary remedial action to be taken, including adjustment to the activity generating the impacts and adjustment to the mitigation or compensation measures.

6.26 The EcIA should identify where monitoring is required for mitigation, compensation and enhancement measures. It should set out the methods to be used, the criteria for determining success/failure, appropriate timing, mechanisms for implementation, frequency and duration of monitoring, and frequency of reporting.

6.27 Monitoring should be secured through a planning condition or obligation built into legal agreements, which the proponent must implement fully (Chapter 7). Monitoring may be used to determine:

- whether the measures have been implemented as agreed
- the success/effectiveness of the measures
- early warning of proposed measures which are not proving effective
- how to remedy the situation should any of the implemented measures fail e.g. due to lack of management.

6.28 Where measures are routinely applied and can be relied upon to deliver well tried and tested mitigation, monitoring is unlikely to be necessary, so long as the competent authority is able to enforce the measures. However, where there are uncertainties in predicting the effectiveness of measures, or mitigation packages are novel, monitoring will be required to determine whether the predicted efficacy of the mitigation meets expectations. In the case of major projects, good practice established in the marine and coastal environment has seen the development of legally binding Mitigation, Compensation and Monitoring Agreements (MCMA). These documents clearly define the measures that need to be taken if the project is to be consented.

6.29 It is vital that monitoring has clear indicators of success or failure, set against a suitable baseline. Monitoring needs to have clear aims and objectives to specifically determine the success of the measures, both in the short-term and longer-term. Longer-term monitoring is appropriate where:

- the success or failure of the measures will take longer to assess – such as for some habitat creation/ management measures
- the mitigation is relevant to a feature upon which the effects may vary during the life of the project.

6.30 It is important that there is an agreed robust feedback mechanism to ensure that where objectives have not been met, provision is made for remedial measures and these are implemented.

6.31 Ideally the results of monitoring should be widely shared, to inform the design of mitigation, compensation and enhancement measures of other projects.

7 CONSEQUENCES FOR DECISION-MAKING

Introduction

7.1 The competent authority determines whether the mitigated project:

- complies with legal requirements e.g. for protected species
- meets national and local policy goals and objectives
- requires conditions and legal obligations attached to the consent concerning detailed design, implementation and monitoring of the project.

7.2 The scoping stage presents the first opportunity to make explicit the legal and policy context. Everyone engaged in the process should be fully aware of, and constantly referring back to, the legal and policy context that applies to the area and issues being studied, and the context of the case. Failure to take account of the legal and policy context, and to provide sufficient information to comply with this, may lead to delay and can result in an application being refused or a decision challenged.

7.3 The consequences, in terms of development control/management, will depend on the importance of the ecological features and the significance of effects upon them and the relevant legislation, policy, and guidance.

Legal Implications

7.4 The legal implications arising out of an EcIA should be made explicit and legal advice sought as necessary. Where an EcIA is undertaken to inform an EIA, it is subject to the relevant EIA Regulations.

7.5 The findings of an EcIA are a material consideration in the planning process and other consent regimes. The competent authority must be provided with all the information needed to assess and evaluate the likely significant environmental effects of a project. The competent authority has the duty to consider the environmental information before it reaches a decision regarding the granting of consent. Key aspects of the EcIA report that the competent authority should take into account when determining an application include[118]:

- the soundness of technical content of ecological information including
    - adequate and up-to-date data
    - ecological methods in accordance with good practice
    - departures from good practice made clear
- whether ecological features are likely to be affected and all potential impacts are described adequately
- whether effects are significant and, if so, are capable of being mitigated
- whether the mitigation hierarchy has been applied
- whether it is adequately demonstrated that the proposal will deliver stated outcomes, with regard to likely effectiveness and certainty over deliverability
- whether the measures are capable of being secured through appropriate planning conditions and/or obligations, and/or are likely to be permitted through other consent regimes e.g. licences for Protected Species
- whether the proposals are compliant with statutory obligations and policy
- whether there is clear indication of likely significant losses and gains for biodiversity
- whether any material considerations have been identified that might require changes to the application.

7.6 When determining a planning application, the competent authority should not adopt a 'wait and see' approach. It should not impose a condition requiring further work to identify the likely environmental effects after permission has been granted. It is a well-established principle that the planning authority should ensure it takes account of all material considerations before making its decision (and does not take account of immaterial considerations not relevant to planning). It is therefore crucial that all information about the potentially significant ecological effects of the proposal is available to the planning authority before it grants permission.

7.7 Where the competent authority considers that the information is insufficient it can request further information or evidence to verify the information already provided. Working closely with the decision-maker, statutory bodies and other consultees during the assessment process should help reduce the likelihood of the competent authority needing to do this.

7.8 In the event that uncertainty cannot be resolved, a compelling case may remain for taking forward a project. In these circumstances, a decision has to be taken in the knowledge that there is uncertainty. In such cases, adaptive management, linked to a bespoke monitoring programme, is recommended as a way of ensuring that any negative impacts or effects are addressed in the course of the development and during the subsequent operational phase. This process depends upon ongoing engagement by the statutory bodies and upon regular reporting by the developer and their consultants.

Policy Implications

7.9 All parties engaged in EcIA should be familiar with the national and local policies that are relevant to a project. Key national policy documents that should be considered during EcIA can be found on government websites and local policy documents are usually available on local authority websites.

7.10 If the project being assessed has emerged from the process of preparing a development plan for which a Sustainability Appraisal (SA) or Strategic Environmental assessment (SEA) has been undertaken, the results of the SA/SEA may be relevant to the EcIA. SA/SEA requires the explicit consideration of alternatives to the project, biodiversity objectives and outcomes. These results should be taken into account along with any mitigation measures, recommendations and monitoring.

Implications for Detailed Design and Implementation

7.11 Conditions, planning obligations and legal agreements are often needed to secure and enforce the implementation of mitigation, compensation and enhancement measures outlined in an EcIA. These obligations can be enforced by the competent authority. This can be particularly challenging when the obligations were developed with one organisation but a different organisation implements the consent.

7.12 All parties should understand the actions they need to take during the implementation stages of a project. This will mean identifying and designing in detail the measures necessary to avoid, mitigate and compensate negative effects, and any measures necessary to achieve enhancements. Details of these measures will be set out in the EcIA and may be presented in an Environmental Management Plan (EMP) or similar document.

7.13 The project proposer should demonstrate commitment to the package of mitigation, compensation and enhancement measures. This is most effectively done by proposing to enter into a legally enforceable agreement, such as, in England and Wales, a planning obligation under section 106 of the Town and Country Planning Act 1990, consent /permit conditions in Ireland and/or its equivalent provision elsewhere. It would normally be between the proposer / developer / landowner and the planning authority, with input from the professional ecologist working on the project. The relevant instrument can include specific obligations which may be set out in detail in a schedule, and may take the form of a Mitigation, Compensation and Monitoring Agreement (MCMA) in the UK or a Monitoring Programme/Plan in Ireland. This should include a detailed explanation of what is to be done, how it will be achieved, where and when it is to be carried out, and who is responsible for ensuring that works are undertaken as proposed. The information provided should include:

- summary of the impacts to be addressed, with clear description of whether they will be dealt with by mitigation measures or compensation measures, and explicit reference to supporting data to ensure an audit trail
- details of how the proposed measures will be funded
- location and extent of the proposed measures on scale plans
- a timetable for implementation of design options and integration with phases of development e.g. construction, operation, habitation, decommissioning, restoration

- expertise of persons responsible for implementing design options
- availability and security of land to implement the design options
- a description of all other resources required to implement the design options
- a statement of how design options will be secured within the planning process or consent process
- details of proposed liaison with local experts and how local people can contribute ideas
- supervision during works by an Ecological Clerk of Works or Project Ecologist
- a monitoring scheme to evaluate the success of mitigation measures and/or compensation measures
- remedial measures in the event that mitigation measures and/or compensation measures are unsuccessful or there are unforeseen effects
- proposed auditing/reporting and publication procedures.

7.14 Good practice in major or novel development projects (e.g. port developments) has seen the development of 'Regulators Groups / Committees' that meet at intervals to discuss progress and the outcomes of monitoring that has been undertaken. This sort of arrangement is best suited to major projects that take place over several years. It could be adapted to smaller projects to ensure that there is a formal completion to the project and to the delivery of compensation, mitigation and monitoring associated with the project. Regulator groups are particularly useful for overseeing adaptive management (see 7.8).

7.15 A clear audit trail should be maintained by both the project proposer and the competent authority in relation to the rationale and timescale for mitigation, compensation and enhancement measures and their delivery and monitoring. This is particularly important where the final project is completed by a successor body inheriting these responsibilities.

APPENDIX 1

Example Assessment of the Significance of Effects: Impacts of road widening on a population of Cetti's warblers

Note: The worked example provided below, whilst intended to be as realistic as possible, is fictitious and for illustrative purposes only. Its aim is to show the general principles of how the approach to EcIA recommended in these Guidelines might be applied in practice. The actual values of various parameters and the conclusions reached could differ in a real situation; other parameters could be relevant and other research findings brought to bear.

The scheme proposals and key biophysical changes

The hypothetical road scheme involves widening a 3 km stretch of two-lane road (c. 9.3 m wide) into a dual carriageway with hard strips (final hard surface increase to c. 19.1 m). This scheme is being implemented largely to take traffic out of the centre of a country town. The widening will require the removal of improved grassland in this location. The scheme is located within the county of Cymrent in the west of Britain and is aligned north-south. Once commissioned, the scheme would permit an almost three-fold increase in traffic volume from 9,500 to an estimated 28,000 vehicles per day and an increase in average speed from 80 kph to 100 kph (c. 50 mph to 60 mph). Site clearance would occur over a six week period, and it is intended to complete the construction works within eight months.

Description of ecological features

- The feature being assessed is a local population of five breeding pairs of *Cetti's warblers Cettia cetti.*
- Approximately 50 m to the east of the A road is a 25.5 ha County Wildlife Site (CWS) called 'The Cuts', which runs parallel to the A road for some 400 m and some 640 m away from it to the east. 'The Cuts' supports five pairs of Cetti's warblers (as well as a diverse range of other species including whitethroat, willow warbler in notable numbers and occasional uncommon migrants, which are not considered further here but which might experience similar impacts to the Cetti's warblers). 'The Cuts' comprises six marshy grassland fields, demarcated by ditches and hedgerows on low bunds. These hedges consist of dense stands of blackthorn, bramble, common alder and grey and goat willow. Common nettle and great hairy willow-herb occur densely adjacent to the bunds. There is a 0.5 ha area of open water and a 1.5 ha reedbed at the centre of one field, but due to natural succession this zone is rapidly being invaded by scrub. The territories of the Cetti's warblers are localised along the bunds, extending typically for lengths of up to 400 – 500 m (in four of the five cases along two or three sides of a field) the birds defending areas of hedgerow, ditch and tall marshland up to c. 20 m either side of the mid-point of the bunds. This average territory size is in keeping with that found in other sites in Britain, though there is high variability (see Snow and Perrins 2000). According to the local Wildlife Trust, territories extend neither into the easternmost c. 150 m of 'The Cuts', where the ditches are dry due to the over-abstraction from local watercourses and boreholes for agriculture, nor into the westernmost 100 m or so (nearest the existing road).
- The Cetti's warbler was unknown in the Britain until 1961. Breeding was first recorded in Kent in the early 1970s. This is a species on the northern edge of its European range that is expanding fairly rapidly in population size and geographical extent (northwards) in response to the recent series of relatively mild winters. It is thought susceptible to very cold winter weather and its population in the mid-eastern counties of England was eradicated by some of the cold winters in the 1980s. The current population size is estimated at over 850 breeding pairs (based on numbers of singing males in 2002), two thirds of this population being in the south-eastern counties of England. Wales was colonised from 1980. The population in Cymrent has been estimated at 50 pairs (singing males).
- Male Cetti's warblers advertise territory with loud song sung from deep cover. Recent research has shown that these songs are highly individual and that recognition of specific neighbours occurs.
- 'The Cuts' is accessed by a footpath usable by the landowner and permit-holding members of the local Wildlife Trust and there is a bird watching hide near the reedbed.



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HAMLYN SPECIES GUIDES

THE MARSH HARRIER

Roger Clarke

HAMLYN

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In the rare instances when tree-nesting does occur, the nest is usually built on a bushy canopy.

Foraging

Marsh Harriers prefer a diversely structured environment, with variation in vegetation height and features such as ditches which offer them the most chances for surprise as they fly over these and suddenly appear at close quarters to prey hidden in them. They therefore tend to avoid large, uniform areas without tall vegetation. The hunting-niche separation of sympatric harriers has been extensively studied in Holland by Schipper *et al.* (1975) in winter and by Schipper (1977) in the breeding season. A clear difference was shown in the height of vegetation preferred for hunting both between species of harrier and between sexes of the individual species. Female Marsh Harriers preferred hunting over wet reedbeds, and males hunted there, too, but they exploited adjoining habitats with shorter vegetation more than the females did.

The habitats surrounding reedbed nest sites are all-important, with farmland species featuring heavily in the diet in some studies. In Flevoland, the dry reedbeds could provide few marshland bird prey, so a preference was shown for hunting in crops (Schipper 1977). Shifts in hunting-habitat preference from reedswamp to arable farmland can take place once there are young in the nest if the arable habitat provides a higher return. Mixed cropping gives even more options. In Norfolk, Sills (1984) found that the pea crop preferred as hunting habitat was far richer in insects than other crops and attracted the juvenile Starlings featuring heavily in the harriers' diet.

Roosting habitats

Marsh Harriers roost predominantly in reedbeds, but, in breeding areas where these are small, autumn communal roosts form in crops. Barley appears to be preferred, but other cereals are used, as are beans, potatoes, sugar beet and other crops. On the coast, alternative roosting habitat is on saltmarsh.

In winter, the presence of other species of harrier in good numbers often seems to draw Marsh Harriers into dry habitats such as grassland for roosting purposes, and even onto bare ground. I regularly saw several Marsh Harriers at a roost predominantly of Montagu's and Pallid Harriers in dry grassland in Gujarat, north-west India. The roost peaked at 2000 birds, the largest harrier roost recorded in the world in recent times (Clarke in press b).

Meinertzhagen (1956) reported a group of 66 migrant male Marsh Harriers roosting together for one night on ploughed land in Afghanistan. In eastern India, Donald (1905) witnessed a roost of 'crowds' of Marsh, Pallid and Pied Harriers on bare ground. He concluded that they were not passing migrants, as they scattered in all directions in the morning.

On 22 January 1993, I was taken to a communal harrier roost in the Little Rann of Kutch, a desert in Gujarat, north-west India. The roost site was 1–2 km inside the desert, where the ground is flooded by the sea in the monsoon, but dries out from about September into a flat surface of cakes of salty mud. Arriving at the roost six minutes after sunset, we counted 11 harriers already standing on the desert; 20 minutes after sunset there were at least 20, but our position was rather too distant from the main concentration to make a satisfactory final count. The following evening we positioned ourselves more centrally and our final count was 30 harriers: about 20 roosted in one main group spread over about 500 m of desert, with several outliers, mostly in a small, loose sub-roost a further 500 m away. Those close enough to be identified by telescope were as follows: Montagu's Harriers – 3 adult females, 7 adult males; Pallid Harrier – 3 juveniles, 1 adult male; Marsh Harrier – 7 adult females or juveniles. This roost was not just a temporary phenomenon, since our guide, Anil Mulchandani, had found similar roosts in the area both the previous winter and earlier that same winter. In the absence of vegetation in which to hide, the harriers clearly must rely on their hearing to protect themselves at night. The dried mud had in many places formed a crust of small 'tiles' with raised edges and separated by cracks;a human foot spanned two or three and produced a crackling sound as weight was put onto it. Mammalian predators must also find it difficult to walk silently on this

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BREEDING BIOLOGY

Age of first breeding

Some immature Marsh Harriers remain on the wintering-grounds in Africa or India in summer. Others colonize unoccupied or newly developed breeding habitats and some fill vacancies in the adult breeding population. In many raptor species, females are recruited into the breeding population earlier in life than males. Reliable information on the ages of breeding female Marsh Harriers is difficult to obtain by observation alone, since their plumage can be difficult to age. With males the position is clearer: full adult plumage is reached in their third or fourth calendar-year, and the age of younger birds can be gauged by their plumage (*see* Chapter 2). Although many sub-adult males are predominantly brown above, their light underwings are soon obvious, and it is uncommon to find a male Marsh Harrier beginning to breed in true juvenile plumage with mainly brown underwings. However, this does occasionally happen, probably mostly at new sites where there is no competition from established males. For example, the male of the first pair to breed in Cambridgeshire in the twentieth century was in brown plumage and gradually moulted into dingy male plumage during breeding. Several young males have been noted breeding in the initial stages of the new semi-colony on the Isle of Sheppey in Kent (P. Hadrill, pers. comm.). As a general rule, older males return earlier to the breeding-grounds, begin the breeding cycle earlier in the year than immature birds, and have greater success (Altenburg *et al.* 1987).

The age structure of the Marsh Harrier summering population seems to vary between the north and the south of Europe. In the Camargue, Thiollay (1970) recorded that, of a sample of 330 observations in spring, only about 28.5 per cent were of adult birds. He estimated that about 30 immatures were present in the area containing up to seven or eight breeding pairs, a predominance which may be more typical of marshes in southern Europe. Altenburg *et al.* (1987), who studied the evolution of a population which began to colonize Lauwersmeer polder in the Netherlands in 1969, found that when habitat first became suitable for breeding younger males predominated, but not in later years. They assessed that up to 20 non-breeding Marsh Harriers in immature plumage were present throughout the breeding season with up to 72 pairs which were breeding, indicating that about 12 per cent of birds present were non-breeding immatures. In Poland, Witkowski (1989) noted only 1–3 unmated territorial males in subadult plumage and up to five unpaired juvenile females among 30–50 breeding pairs. Observations in one well-recorded area in east England in 1988 suggested that about 34 per cent of the summering population were non-breeders

there (Rare Breeding Birds Panel). For Britain, over the period 1947–80, Underhill-Day (1984) found a highly significant correlation between non-breeding summering birds and the number of breeding attempts in the following year. This may be expected in a recovering population.

Timing of the breeding cycle

The Marsh Harrier begins to nest earlier in the year than other harriers. This follows the general rule among raptors that larger species begin their longer breeding cycles earlier.

They return to breeding areas north of the wintering range from mid March to early May, depending on latitude (Glutz von Blotzheim *et al.* 1971). On the east coast of England individuals take up territory from March. The passage of migrants brings new birds into sites until mid May and stimulates territorial behaviour. In western Poland, Witkowski found that the return of males peaked at the end of March, but that of females in mid April. Bengtson (1967) in Sweden found that males usually return ten days before females. Spring migration statistics from the Mediterranean show that males migrate significantly earlier on average.

Courtship commences immediately the birds return, and the nest is normally built in about 2–3 weeks. Incubation begins with the first egg and lasts about 33 days. The young hatch asynchronously and stay in the nest for about 28 days. They are brooded continuously by the female for the first few days but gradually become more mobile and venture into the surrounding vegetation, returning to the nest to be fed. From about 28 days, the young can fly, but mostly sit about in the vicinity, typically on bushes; from about 45 days they fly around more but are still fed; from about 73 days they are more independent.

A clear relationship can be shown between latitude and the mean date of commencement of the clutch. In Spain and the south of France, many pairs begin laying in early or mid April. In England, the Netherlands and Poland, all at about the same latitude, the second half of April is the normal time, and farther north, in Scandinavia, clutches are begun in the second week of May. There also appears to be a differential between coastal and inland sites. I have suspected this from casual observation at the few inland nests in England, and Hildén and Kalinainen (1966) measured this in Finland, where the mean first laying date was 9 May on the coast, but 14 May inland. Schipper (1979), Witkowski (1989) and González (1991) all established that laying date was influenced by temperature. Laying began a few days after a rise in temperature, but only at the right time of year and if there was not a subseqent sharp cold spell. The rule put forward by Schipper (1979) for the Netherlands was as follows: 'The first date after 9 April with a temperature of over 4 °C followed by at least two days of temperature rise, whereas on the third or fourth day the temperature should not fall more than 28 per cent.' Apart from the obvious benefits of laying eggs in warmer, more settled temperatures, the key to this rule probably lies in the sychronization of breeding with the peak availability of young of waterfowl and other prey.

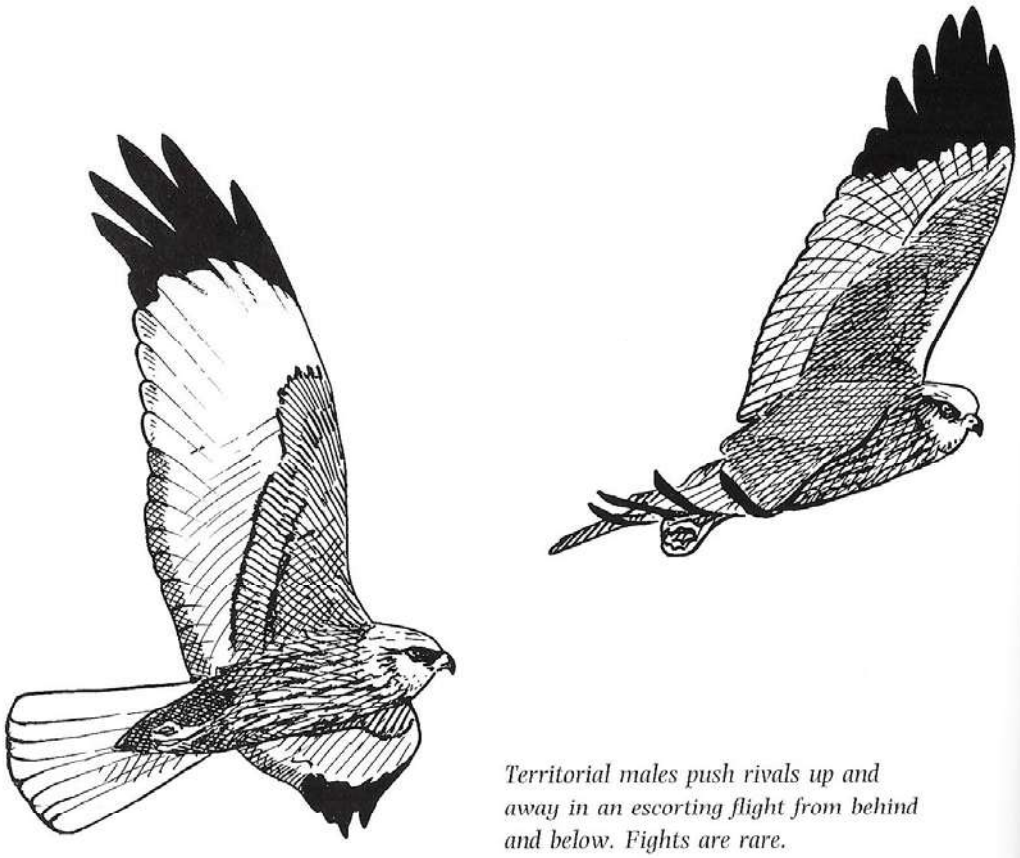
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TERRITORIAL BEHAVIOUR

The numerical strength of the Marsh Harrier breeding population pivots on the capacity of available marshland, a scarce resource. Spatial strategy is therefore highly important for this species. Harriers in general tend to nest in clumps or 'semi-colonies' but hunt individually, ranging out extensively over surrounding land. This arrangement effectively maximizes the capacity of scarce nesting habitat. The Hen Harrier and Montagu's Harrier also breed in marshes, but not nearly as exclusively so as the Marsh Harrier, and the tendency of Hen Harriers, for example, to nest in semi-colonies in one area of a wide tract of seemingly uniform moorland habitat indicates that there may be some additional factor to clumping. The basing of non-breeding activities on a communal roost site is a similar spatial arrangement used by most harriers, but winter communal roosts are also known in solitarily nesting species of raptor.

Breeding density seems to be less in the Marsh Harrier than in the smaller Hen and Montagu's, perhaps as a result of its larger size and more restricted habitat preferences which limit numbers and require greater spacing of nesting pairs, especially in the early stages when the birds spend more time in the breeding marsh. Breeding density, however, can be as high as a mean inter-nest distance of 100 m (Berg and Stiefel 1968), in effect a loose colony. Witkowski (1989) recorded a mean annual density of down to 26.5 ha per pair within the area used by Marsh Harriers for nesting, with a mean distance of 210 m between nests in the wider reedbeds along the shore. In the Czech Republic and Slovakia, Danko et al (1994) found that marsh habitat was used very intensively in dry years, with nests often only a few tens of metres apart.

During the courtship and incubation periods, activity is concentrated on the breeding marsh. Territory size is at this stage typically small – a radius of 100–300 m around the nest – and the male spends a lot of time in early spring patrolling this and defending it against all perceived enemies such as crows and predatory mammals. The defence of territory against rivals varies in intensity, eliciting responses varying from threatening to escorting and to attacking. Moles are the most frequent defenders of territory against other Marsh Harrier but they are more tolerant of female intruders early in the season (Fernández and Azkora 1994b). Behaviour involving actual contact is rare. Escorting takes the form of following, chasing or 'pushing' the intruder up or away out of territorial airspace. Flying with legs lowered is a form of threat used to maintain territorial boundaries and if possible to avoid potentially damaging contact. This was reviewed in various territorial predatory



Territorial males push rivals up and away in an escorting flight from behind and below. Fights are rare.

birds by Barnard and Simmons (1986). Occasionally, this demonstration will escalate into a fight, and, as in play, an attacked bird's reaction is to flip over and present talons, and rarely the two will actually grasp each other by the talons and fall to the ground. Observing this in the African Marsh Harrier, Simmons (1991a) recorded that 'these encounters lasted 1–7 minutes, while the birds were presumed fighting or locked together on the ground'. At Horsey, Buxton found that some male Marsh Harriers were particularly pugnacious and one year he found the corpse of the first male to arrive in spring floating in the dyke under its preening post, 'no doubt slain by his rival who apparently lost a wing feather in the process'.

Once the young are hatched, the males abruptly extend their activities beyond the nesting territory by making long foraging flights into surrounding, often marginal or arable, land. The reason for this is very clear. Unless a particularly rich source of waterfowl is available, marshland hunting is not very productive. At Titchwell in one season, Sills (1983) observed that it took a male an average of 27 minutes to catch prey in the marsh, and it was likely to be a small item such as a fledgling passerine; prey was caught every 17 minutes on average on farmland and was usually a much heavier young Pheasant, Rabbit or Starling. Sills calculated that farmland was ten times more productive in terms of weight of food per minute of hunting. Elsewhere in East Anglia, Underhill-Day (1989) established that the size of

the hunting range of a male was very much larger during the nestling stage than it was earlier or later.

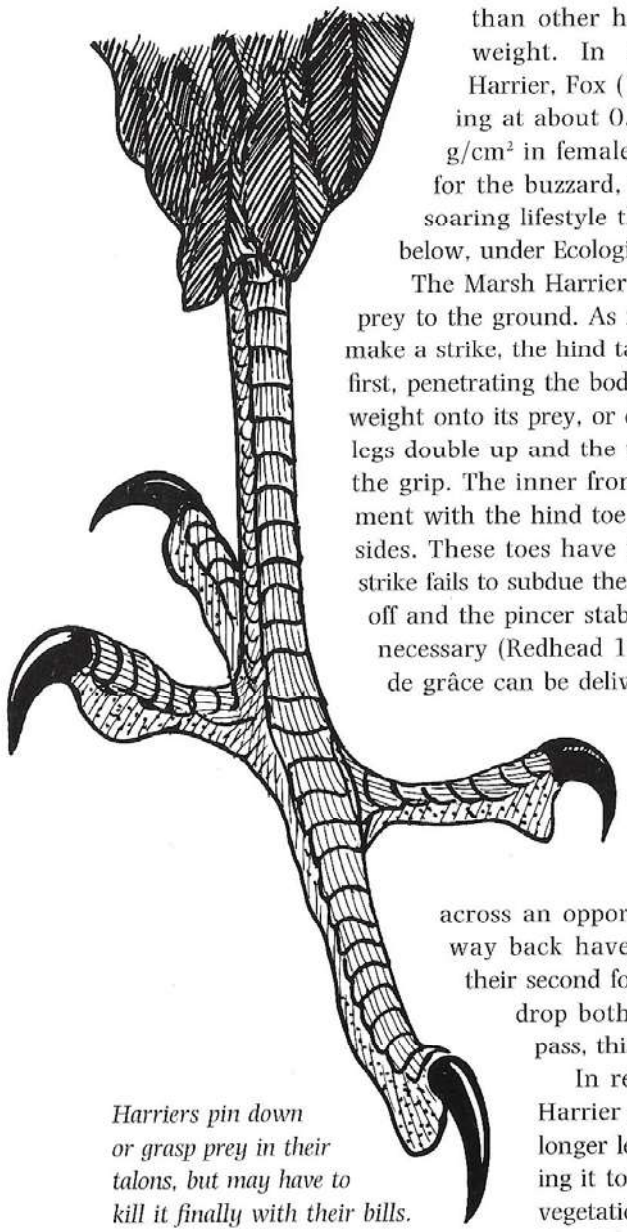
The males tend to hunt sectors of the compass based on their nest. Some researchers have found that such hunting territory is defended against other Marsh Harriers (Sondell 1970). Others maintain that it is not unusual for the hunting ranges of neighbouring males to overlap and that hunting territories are not defended.

How far might a harrier range from the nest? Maximum hunting distances from nests by male Marsh Harriers were recorded by Schipper *et al.* (1977) in Holland and France at only 1.5 to 3.1 km, females ranging only 1.4 to 1.8 km; they quoted other researchers as recording 5–8 km. In Cambridgeshire, where nesting pairs have had no near neighbours, I had one observation of the Wicken Fen male well away from the nest: one lunch-hour in 1985, at about the end of the harriers' incubation period I spotted him coursing low over the reeds of a marsh 12 km south of Wicken; he soon circled up in the sunshine and made off south, away from Wicken! There was no doubt that it was he – the same two inner primaries missing with no sign of regrowth and one or two tail feathers cut off short (shot damage acquired on migration?). On a separate occasion he was seen at another marsh 10 km east of Wicken.

The sizes of hunting ranges quoted by Schipper *et al.* vary from 250 to 680 ha for males and 80 to 370 ha for females. At Titchwell, hunting ranges out over arable farmland, inland of the coastal breeding marsh, were measured for individual males in 1982 and 1983 at a much larger 1250 ha and 1000 ha respectively (Sills 1988). Elsewhere in East Anglia, Underhill-Day (1989) calculated 217 ha during the courtship stage, 170 ha during incubation, 1112 ha during the nestling period and 310 ha during the fledgling period. In Poland, Witkowski (1989) calculated a far smaller mean area of 125 ha of arable fields and meadow plus 35 ha of fishpond used by each pair in a large semi-colony, with up to 50 pairs using 80 km^2. It seems clear that, so far as range sizes go, it all depends on the habitat, prey density and the stage of the breeding cycle.

Adaptations

The light structure and long wings of the harriers are adaptations to prolonged searching on the wing. Wing-loading measurements (weight of bird divided by wing area) for the harriers are much lower than those for raptors of similar size other than the kites, which are also very aerial birds. Harriers range from about 0.21 g/cm² in the smaller species to 0.30 g/cm² in the Marsh Harrier (Brown and Amadon 1968). Even a relatively heavy harrier such as the Marsh Harrier has a far lower wing-loading than, for example, the Common Buzzard at 0.45 g/cm². The Marsh Harrier has a shorter tail and smaller flight apparatus than other harriers, in relation to body-weight. In the New Zealand Swamp Harrier, Fox (1977a) calculated wing loading at about 0.39 g/cm² in males and 0.41 g/cm² in females, values approaching those for the buzzard, whose less active and more soaring lifestyle they were converging on (*see* below, under Ecological release and change).



Harriers pin down or grasp prey in their talons, but may have to kill it finally with their bills.

The Marsh Harrier normally kills by pinning its prey to the ground. As it pounces and reaches out to make a strike, the hind talon or talons generally arrive first, penetrating the body of the prey. Falling with its weight onto its prey, or clutching it to it, the harrier's legs double up and the tendons inside pull to tighten the grip. The inner front toe acts in a pincer movement with the hind toe to pierce the prey from both sides. These toes have the longest claws. If the first strike fails to subdue the prey, then a foot can be lifted off and the pincer stabbed in again, several times if necessary (Redhead 1969). Alternatively, the coup de grâce can be delivered with the bill, but not as a matter of course as the falcons do. In the breeding season, prey is generally carried back to the nest secured in one foot, and harriers that come across an opportunity to kill again on their way back have been known to strike with their second foot. If a male then proceeds to drop both prey to a female in a food-pass, this can cause problems!

In relation to its size, the Marsh Harrier generally has larger feet and longer legs than other harriers, allowing it to reach down into long marsh vegetation for prey. The one exception

is leg length in eastern Asia of the Pied Harrier, which proportionately exceeds that of the Eastern Marsh Harrier there and is also thought to be an adaptation for reaching into deep marsh vegetation (Nieboer 1973). The large feet of the Marsh Harrier indicate greater killing capability for its body size. Comparison of the foot size of a male Montagu's Harrier and a female Marsh Harrier demonstrates the range of capabilities within a genus of birds which appear relatively similar in size in the field. This is a clue to their niche separation by size of prey.

The harrier bill reflects no special adaptation. It is again larger in proportion in the Marsh Harrier. This enables it to deal better with the larger prey its larger foot size allows it to catch, and with prey relatively difficult to tear or break open, such as fish, amphibians, reptiles and crustaceans. It also allows it to compete better at carrion. A side view reveals a very pronounced protruding hook to the upper mandible, which is used for tearing food.

Harriers hunt both by sight and by hearing. The forward-facing eyes allow good judgement of distance at the moment of pouncing on prey and major anatomical adaptations have enhanced the directional quality of their hearing, probably making it as important as their eyesight. It is this which determines much of their distinctive lifestyle and makes them harriers. For birds of their size, they have very large ear openings, set in facial discs. The supraorbital ridge forms the effective top to the half-disc, giving the face a fierce expression from some angles. The ridge may also protect the eye from glare, or when attacking prey. The external structure of the facial disc is very specialized. The ruff curves around just behind the ear opening and below the cheek. It is made up of a ridge of skin set densely with stiff perpendicular feathers, curved back with the general lie of the plumage only towards the end of their length. Highly specialized in form, the barbs are very dense and are knitted together by abundant barbules, giving these feathers a silky appearance which reflects light and makes the ruff conspicuous. In the male Marsh Harrier plumage, they are less pigmented and stand out all the more. Lifting the ear-coverts of a Marsh Harrier reveals an oval ear opening approximately 12 mm long and 9 mm wide. The opening begins about 15 mm behind the eye and the forward rim is raised, meaning that sounds tend to enter from behind, by reflection off the ruff of feathers. The ear-coverts lie back flat over a bare cheek and the ear opening, and are also finely adapted, with well-spaced barbs and blunt barbules, with the result that the vane does not knit together as in any normal feather, thereby allowing sound waves to pass through easily whilst still preventing the admission of particles of dirt borne on the wind. The feathers of the various parts of the disc are independently mobile, with their own musculature and nerves, allowing the harrier to direct sound into the ear by lifting feathers and altering the shape of the disc.

So, how exactly are these anatomical features used? In the owls, similar adaptations are for pinpointing prey in darkness, whereas in the harriers they are for pinpointing prey hidden by rank vegetation. The owls differ in that their ear openings are positioned asymmetrically and enable them accurately to gauge the source of a sound not only on the horizontal plane, by turning their head to equalize the amount of noise in each ear, but also on the

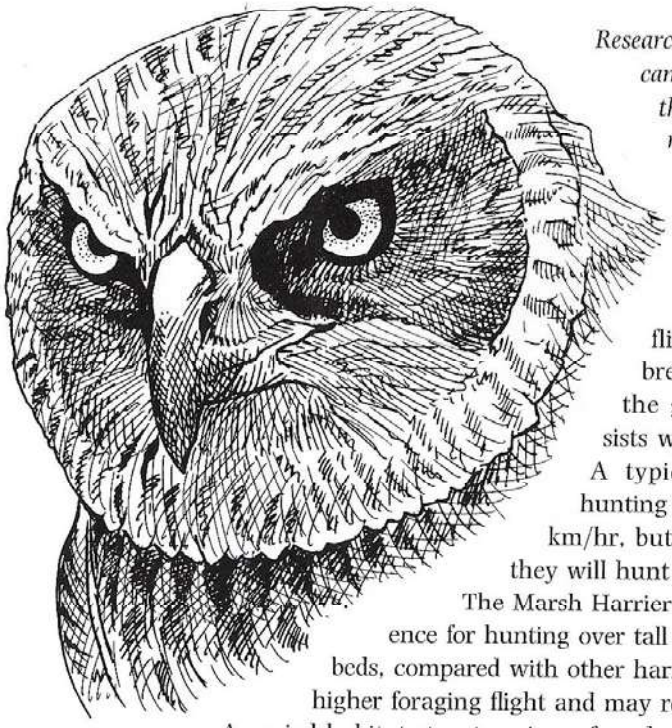
vertical plane. The lack of asymmetry in the harriers does not drastically cut down their directional hearing ability, however, and it may not be that critical, since in many situations the harrier can use its eyes at the last moment as it appears suddenly over the vegetation, on the correct horizontal course for the prey. An American scientist, William Rice (1982), tested the hearing of the Northern Harrier in laboratory conditions and found that it was almost equal to that of Short-eared and Barn Owls on the horizontal plane. Also working in the laboratory, Van Dijk (in Nieboer 1973) confirmed that the hearing abilities of Montagu's, Hen and Marsh Harriers were better than those of Common Buzzard, Goshawk and Kestrel. The Marsh Harrier therefore has these listening adaptations, but they may be less pronounced than in other harriers and the Marsh Harrier clearly must use them less because it tends to fly higher, giving better vision into tall vegetation but increasing the range for hearing purposes.

It would appear that the harrier's facial discs gather sounds in much the same way as a parabolic reflector. Recent laboratory research on the coding of sound frequency and location in the brain of the Swamp Harrier, however, has found that the upper limit of response is to tones of about 4 kHz (Calford *et al.* 1985). Calculations of the point at which a facial disc of the size of the harrier's would become directional suggest that this would start only at about 4 kHz, meaning that the disc is too small to have a significant effect on the sound field in the harrier's audible frequency range. Although perhaps too much should not be read into this until it is confirmed by other studies, further research should prove very interesting indeed. The harrier's adaptations with regard to the facial disc and ruff are so similar to those of owls that it is almost inconceivable that they have a different function, but research has already found that the owls' methods of processing sounds in the brain may be quite different.

The sexual dimorphism in plumage of the Marsh Harrier may be significant from a hunting point of view. The much paler underside of the male may give better camouflage against the light sky in the more open situations where he encounters prey. Such camouflage could be less important for the female, as her hunting technique relies more on outright surprise in the more structured habitats in which she prefers to hunt.

Foraging techniques

Marsh Harriers typically fly over likely terrain at 2-6 m above ground level, cold-searching for prey beneath them. The height of flight seems to be linked to the height of the vegetation (Schipper 1977), with taller vegetation requiring higher flight to reduce the angle of vision for the harrier to be able to see down into it. Harriers use a low approach, screened by the vegetation or some other low obstacle. The technique is geared to surprising and closing on prey effectively at the first encounter. Marsh Harriers are not well adapted to give chase; they will go after running prey in the open with some expectation of success, but rarely pursue flying prey. The harriers' typical mode of flight, flapping for a few wingbeats but then gliding may be significant. This produces a rising and falling cadence to their



Research has shown that harriers can pinpoint prey solely from the noise it makes. The stiff ruffs of feathers behind the owl-like facial disks direct sound into large ear openings hidden by flimsy covert feathers.

flight, especially in a stiff breeze, and it may be that the glide on still wings assists with stealing up on prey. A typical flight speed for a hunting Marsh Harrier is 20–30 km/hr, but in some circumstances they will hunt more slowly.

The Marsh Harrier shows a greater preference for hunting over tall vegetation such as reedbeds, compared with other harriers. It has a noticeably higher foraging flight and may rely more on visual cues. A varied habitat structure is preferred, with abrupt changes in height of vegetation and other obstacles, giving more opportunities for surprise as they are flown over. Examples are ditches, verges, field edges etc., linear features that are followed by harriers. A harrier may hunt a promising patch of rank vegetation by the more thorough 'quartering' technique, changing direction frequently and flying over the patch several times. It may fly into the wind over part of the habitat, then rising up and quickly gliding back as its wings catch the wind; it then flies back low across the habitat into the wind again while carefully searching for prey concealed in the vegetation. A headwind allows the harrier to remain in control while reducing both its speed, to allow a more thorough search, and its altitude, to allow it to strike more quickly.

The heavier and less agile female Marsh Harrier appears to profit from the greater opportunity for surprise that really tall vegetation such as reeds gives her. Also, as a less manoeuvrable bird, she is not able to react so nimbly to catch prey that she comes upon at very close quarters. Dropping down into reeds that confine prey may give her more latitude, and combined with a slower speed of hunting flight at greater height, reedbed hunting is most typical of the female. Flying into the wind; at times the harrier seems to remain virtually at one spot, clumsily hovering where its interest is aroused. Males are more often seen hunting low and over shorter vegetation. The Marsh Harrier's slower hunting flight, especially that of females, means that it covers less ground than other harriers, but works it more thoroughly. In the breeding season, females hunt mainly on the nesting marsh, but males do tend to range further afield over the other habitats which they are better adapted to exploit.

Often, a Marsh Harrier will test a flock of waterfowl feeding out in the open, leisurely flying at and flushing them, in effect testing for the weak and slow. Hunting for passerine birds requires a fast approach. In Norfolk,



Male hunting over rough grassland. The rank vegetation helps to conceal his approach from potential prey and the light fore-edge to the wing breaks his outline.

flocks of juvenile Starlings feeding out on farmland in late summer are a favourite target. A male Marsh Harrier will fly fast and low towards them, if he is successful seizing one on the ground or in flight (Image 1991). Fast hunting, however, is not commonly used by the Marsh Harrier.

Flying low, harriers often cannot react to prey until they are already passing over it. A 'roll-pounce' as the harrier stalls, spreading its tail and cartwheeling back to dive feet-first at prey is typical; but the strike can also take the form of first hesitating or hovering, and then sideslipping through the air to drop straight down on to the prey, legs outstretched and wings held up. Hesitations in flight/feints occur at some height, but can be quickly checked if the harrier perceives that it has made a mistake or it has no chance of success because the prey has bolted. Stooping at prey and diving to flush it out are rarer, but tactics can become quite cheeky: a Marsh Harrier will occasionally knock a female Pheasant off her feet or even settle on her back, intent on stealing a chick. Harrying of waterfowl such as Coot is common. The waterfowl take to open water at the approach of a harrier and dive if necessary. Defensive behaviour such as grouping, turning on their backs and kicking out has been observed in Coots (Harris 1973). Deliberate holding-down or drowning of smaller waterfowl does occur, Hollands (1984) recounting how one determined Swamp Harrier forced a Coot to dive over and over again until it was exhausted and could be seized. The carcase then has to be retrieved from the water, but it may prove too heavy; in India I have watched Marsh Harriers attempt, unsuccessfully, to lift a waterfowl carcase out of a lake. Hunting prey in the water can also be dangerous, as shown by the drowned Swamp Harrier recovered with its talons embedded in a dead Australian Darter's neck (Vestjens 1972).

It is not generally in harriers' nature to still-hunt in the way that, for example, a Kestrel, Sparrowhawk or Buzzard will – sitting on a vantage point watching for prey and then going for it when it appears. Occasionally, especially where voles are numerous, harriers will 'ground-hunt' – sitting in quite short grass and then leaping on a vole when one appears (Schipper *et al.* 1975). At Minsmere, late one February, King (1961) observed a pair of Marsh Harriers jumping on to tussocks and eating small items in them. On another occasion at Minsmere, Clegg (1961) saw three juveniles attempt

ground-hunting, making short bounds with wings flapping, sometimes making a further hop and landing with wings spread, and appearing to miss small items of food. The description of this activity agrees with Cheke's (1987) description of a brood of young Réunion Harriers on the bed of a drying-out lake taking small unidentified items after a 'hop, skip and jump'. Ground-hunting was recorded by Baker-Gabb (1984) as a hunting method of very minor importance for the Swamp Harrier.

The harriers' cold-searching foraging methods and lack of chasing over long distances or stooping make them generally unsuitable for falconry,

To strike, a foraging Marsh Harrier typically spreads its tail to stall, cartwheels over and dives feet-first on prey.





A harrier may hesitate where it detects a potential prey which has taken refuge in deep vegetation, feinting strikes and choosing the moment to press one home.

although the Swamp Harrier has been successfully trained by traditional falconry methods and flown at Rabbits in New Zealand (Wheeldon 1981, 1992). A lot of effort was necessary, though, including selection of harriers which would concentrate on the matter in hand (found to be females 2–3 years old), training harriers to progress from easy young Rabbits to the adults, and flushing or successfully ferreting out Rabbits from their holes. Wheeldon remarked: 'countless hours were spent in the field, but successes were few'.

Strike success rates

Hostile accounts of birds of prey often give the impression that they easily kill everything they go after, well in excess of their food requirements. In actual fact, studies of hunting in a wide range of birds of prey have come up with rather low success rates, typically 5–10 per cent. Rates are difficult

to measure for the Marsh Harrier because the dense vegetation in which it hunts can conceal the outcome, although the birds usually soon fly up with prey and carry it at least a short distance to a place where they feel comfortable dealing with it. Schipper (1977), from observation of birds in Holland and France in the breeding season, calculated success rates ranging from 5 to 17 per cent; Underhill-Day (in Clarke in press a) calculated an unusually productive success rate of 27 per cent by six males and one female in the breeding season in East Anglia. Hunting activity is normally reduced or curtailed by rain. This and the effects of wind speed and temperature on success remain to be fully quantified. Schipper (1973) established that rain and stronger winds reduced the rate of prey supplied to the nest.

Kleptoparasitism

Marsh Harriers have stolen a fish from an Osprey (Persson 1971), a Teal from a Peregrine (Williams 1995) and robbed Hen Harriers of prey (Schipper 1977). At Horsey in 1947, Buxton noted a polygynous male Marsh Harrier stealing prey from a widowed female and a pair of Montagu's Harriers; it 'dashed at the hen Montagu and bagging her prey swept off with it in the direction of his own nests'. Baker-Gabb found that Australian Swamp Harriers were dominant in kleptoparasitic encounters, robbing Whistling Kites and a Peregrine Falcon, and repelling the great majority of attacks by Whistling Kites, Little Ravens and a Brown Falcon.

Monitoring food

In the breeding season, the diet of birds of prey is determined from observations of food brought to the nest or by identifying prey remains at plucking places and in pellets. Older studies sometimes relied on the stomach contents of shot birds.

Identification of the prey species by direct observation from a hide is not so easy as might be thought, because the action can be fast. Also, items brought to the nest may not be whole and a good view may not be obtained. Observation from greater distances may allow approximate classification of prey by size alone. The accurate identification of prey remains in harrier pellets requires skills in identifying principally feathers, fur and rodent teeth, but the greatest problem in the breeding season may be in obtaining enough pellets to give a large enough sample. In winter, harrier pellets can be obtained in good quantities from some communal roosts, but tides or winter floods can wash them away and flooded reedbeds can be impossible for a human to search.

Breeding-season diet

The accounts handed down to us by Buxton of food taken in Broadland indicate some specialization by individual harriers, but show that the commonest prey was Moorhen, with leverets and young Pheasants the next most frequent. Young Moorhens, partridges, Pheasants, buntings, a Blackbird, a Starling and a frog were all found over a period at one food

cache. In other years, adult Grey Partridges were found at one nest, small Coypus at another, and Moles were noted at a third.

Hosking (1943) made a useful record of the prey brought to a nest at Hickling in 1942, although he regarded it as incomplete since smaller items may have been immediately swallowed whole and gone unrecorded: 71 young birds comprised Pheasants (29), Grey Partridges (21), Red-legged Partridges (8), Meadow Pipits (7), Moorhens (2), Mallard (1), Lapwing (1) and unidentified passerines (2); the mammals comprised Rabbits (15), leverets (7), Field Voles (7), Water Voles (5) and a young rat (1). At Minsmere, Axell (1964) cited Rabbits, very small Coypus, Water Rail and young ducks, Moorhens, Coots, Lapwings, Snipe, partridges and small passerines (Starlings and Yellowhammers).

Recent studies of breeding-season diet in Britain have carried out by Norman Sills and colleagues at Titchwell Marsh, Norfolk, looking at two or three nests each year (Sills 1984), and by John Underhill-Day and volunteers on reclaimed farmland in East Anglia (Underhill-Day 1985). At Titchwell, prey seen to be taken by a male harrier were mainly Starlings one year, owing to the abundance of young Starlings feeding in flocks, and young Pheasants the next, the latter taken in situations where broods had wandered into a field, away from the hedgerow; young Rabbits were also a significant prey. The female took generally larger prey, which she brought to an area of rough, flattened grass for dissection before taking them into the nest. Observers were able regularly to collect remains from the place. A seasonal pattern was noticed with Field Voles taken in May, Moorhens in June

Marsh Harrier pellets from a winter roost on saltmarsh.



and July, and young Rabbits in July. A few partridges, ducklings and passerines, notably Skylark, were taken.

Underhill-Day's study showed a wide variety of prey; adult small passerines and mammals (mostly Field Voles and Rabbits) were predominant in the diet in April–May, but young gamebirds (mostly Pheasants) and waterbirds (mostly ducklings and Moorhens) increased in importance thereafter

In the breeding-season diet elsewhere, Schipper (1973) found that the important categories were young waders and Pheasants, and the larger mammals in Holland, but largely passerines with some waterbirds and larger mammals in northern France, and snakes and larger mammals in southern France. Observations were made of Marsh Harriers eating the eggs of ducks and gulls, but these were not of course visibly represented at the nest. In the Camargue, Thiollay (1970) found the principal foods to be frogs, fish and young waterfowl; eggs, reptiles, small mammals, rats, young Rabbits and, to a lesser extent, small passerines also featured. In Spain, González (1991) found that Rabbits made up 46 per cent of the breeding-season diet, with the rest spread over many categories, none of them approaching Rabbits in importance. Work in Poland, on three pairs of Marsh Harriers nesting at a Coypu farm, showed that the males fed mainly away from nesting marshes on rodents and open-country small passerines, whereas females fed mostly in marshes, on the birds there (Pinowski and Ryszkowski 1961). Predation on young Coypu was found to be negligible, as most hunting took place early in the day whilst Coypus were active in the evening. Witkowski (1989) recorded a significant amount of dead carp (10.5 per cent by number) in the diet at a complex of fishponds in western Poland, Common Vole (14.4 per cent) and Coot (12.4 per cent) among a very wide range of birds (60 per cent), mammals (28 per cent), fish (10 per cent) and frogs (2 per cent); eggs were important early in the season, but could not be quantified. Again, the females hunted the ponds and marshland for large prey and the males concentrated on the meadows and arable fields for smaller prey. In the post-fledging period, as the young of marshland prey species grew up, attention was shifted more to farmland for hunting, especially for voles. In Schleswig-Holstein, Bock (1978) found that the numbers of a wide range of largely small passerine birds taken fell dramatically in a good vole year in favour of the Common Vole. Other prey were negligible in importance, but again a taste for eggs was noted.

Witkowsi (1989) described how the shells of eggs eaten by Marsh Harriers were left with a small hole and claw marks. Egg-eating was actually witnessed by Tom and Pam Gardner (1984) in Australia, where Swamp Harriers were prospecting an ibis colony and a harrier dropped on a temporarily unattended nest, mantling the eggs and scaring neighbouring ibises away. Opening its gape wide, the harrier was able to pick up an egg and drop it on the others to crack it; it then opened it up with its bill and ate the contents. It treated all five eggs similarly and then did the same to a nearby clutch of four eggs, before wiping its bill on the nest and flying off.

These results demonstrate the wide spectrum covered by the breeding-season diet of the Marsh Harrier, tuned to the most profitable prey vulnerable at the particular place and at the time.

Access management

Although nest sites in reedswamp can be protected from human access relatively easily, the effect of open access to surrounding habitat is to reduce the usable hunting ranges for the harriers. This can be particularly detrimental if the areas affected contain optimum foraging habitat, and also later in the breeding season, when tourism increases at the same time as the food requirements of Marsh Harrier young reach their peak. In a study on the Neusiedler See in eastern Austria, which has one of the most important Marsh Harrier populations in central Europe, the harriers would fly no closer to visible tourist activity than an estimated 90 m in the open; routes frequently used by cyclists and walkers created a no-go corridor for Marsh Harriers 240 m wide, severely limiting the prime foraging habitat available to them (Gamauf 1993). Appropriate screening of paths and other areas of open access is therefore of paramount importance in reserve design. At Titchwell Marsh, a very heavily visited RSPB reserve in Norfolk with unscreened paths alongside the freshwater and tidal reedbeds, Sills (1983) found that Marsh Harriers would not build closer than 450 m to a path and on average sited their nests 600 m from the footpath to the west and 800 m from that to the east; this was in response to a regular and at times constant flow of people. Because of the proximity of so many nest sites to sea walls, the Marsh Harrier may be vulnerable to disturbance from leisure developments such as long-distance coastal footpaths.

Protection schemes

Although undoubtedly responsible for the total extinction of the species as a British breeding bird at the turn of the century, persecution is currently not a big problem for the Marsh Harrier in Britain, but it could become more of a limiting factor as numbers increase. In the ten years up to 1988, 22 incidents of persecution were recorded (Underhill-Day in Gibbons *et al.* 1993); most individuals were shot, trapped or were the victims of the indiscriminate use of poisoned baits. Two recovered at Snettisham in north Norfolk in 1985 were suffering from gunshot wounds. The species is prone to the illegal and careless use of poison, since it comes readily to both carrion and eggs, which are the principal vehicles used for poisoned baits. The setting of unprotected poison baits in the open is illegal in the UK. In 1980, the male of the first pair to attempt nesting in Scotland for many years was found poisoned (Buckland *et al.* 1990). Unrecorded persecution in the form of intentional disturbance of birds interested in breeding may be a problem in some areas.

Game interests are concerned with the impact of raptor predation on lowland gamebirds, which do form a proportion of the food of the Marsh Harrier. Calculations based on brood-size counts of Mallard, Pheasant and partridges in one study by Underhill-Day (1989) indicated that many of the chicks taken by Marsh Harriers would have died before reaching maturity. The number fledged was reduced by about 13 per cent for Mallard and 11 per cent for Pheasant. He recorded very few partridge chicks taken, perhaps because they stay closer to cover at field edges than Pheasants do. Delaying release until Pheasant poults are of a good size and the peak food demands of breeding Marsh Harriers



The interface of leisure activities and harrier breeding habitat requires careful management. Marsh Harriers are notoriously prone to deserting their nests.

are well past would be a good management strategy; effective cover for poults would also reduce predation. Calculating the losses of gamebirds to raptor predation entails a great deal of work in recording the large number of variables: food delivery rates in each phase of the breeding cycle, diet composition, hunting-range size, gamebird density, and brood sizes of both predator and prey.

Any prejudice against raptors cannot effectively be met by enforcement of the law alone. The requirements of the species in the modern landscape must be frankly discussed between landowners and ornithologists on the basis of sound scientific data and an acceptable position be reached by means of manipulation of the habitat.

In recent years, about half the British Marsh Harrier population has bred in nature reserves or under some form of wardening scheme. The RSPB has wardened a number of vulnerable sites off nature reserves, especially in west Norfolk (where a few pairs of Montagu's Harriers also breed), and in Somerset, for example, the Hawk and Owl Trust has combined with the county Wildlife Trust to warden Marsh Harriers. Routine farming operations are a threat to crop-nesting harriers, making it necessary to keep track of locations of nests and to liaise with the farms involved. Spraying during incubation appears to cause no harm if the nest is avoided by the tractor wheels! Later, the young have to be temporarily covered or removed or the spray turned off as the boom passes over the vicinity of the nest. Spray drift also has to be considered. Crops often grow unnaturally thick and can smother the nest and young, especially in wet weather; oilseed rape is a particular danger in this respect. The crop may have to be cut back at the nest to give the young a better chance.

The time of greatest danger is at harvest. Marsh Harriers nest earlier than Montagu's, and tend to have fledged by harvest time, but where they have not action has to be taken to protect nest from combining operations. It may be possible to agree a delay in harvesting until the young fledge. Otherwise the young should be temporarily removed in a ventilated cardboard box during

Bat Conservation Trust

Bat Surveys for Professional Ecologists

Good Practice Guidelines



3rd edition

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Chapter 8

Bat activity and back-tracking surveys

8.1 Introduction

This chapter provides information on carrying out bat detector surveys for bats. These bats may be commuting, foraging or exhibiting social behaviour (such as calling for mates during the mating season or swarming in the autumn). Acoustic surveys enable identification of species and provide an index of bat activity. Actual numbers of individuals can often not be established unless acoustic data is coupled with direct observations in the field by an ecologist, or through recordings made by an infrared or thermal-imaging camera.

These surveys may be required where development proposals are likely to impact on habitats suitable for bat commuting and foraging (see Section 2.2.2). Road and rail schemes can cause the specific impact of collision and it is good practice to carry out automated/static bat activity surveys of crossing points.

As with all surveys, survey design should be based around the questions that require answers. For the purposes of development and planning, the main questions with respect to bats in flight away from their roosts are generally as follows:

- Are bats present or absent?
- Which bat species use the site?
- What are the activity levels of bats on the site and can this tell us anything about the abundance (number) of bats using the site?
- What are bats using the site for?
- What is the temporal (both seasonally and in relation to time of night) and spatial distribution of recorded bat activity on site?
- Are peaks in bat activity associated with particular temporal and/or spatial locations, e.g. times of night or parts of the site?
- How are the habitats used on site connected to habitats in the surrounding area?

Answering some or all of these questions should allow an ecologist to carry out a robust impact assessment.

In order to answer these questions, bat activity surveys generally begin with the preliminary ecological appraisal, which includes a desk study and fieldwork (see Chapter 4). This provides existing data about bats in the area and identifies and assesses the suitability of habitats on site for bats. This information should be used to inform survey design, which should be iterative; each stage should inform the next.

The following sections describe **transect** and **automated/static bat activity surveys, back-tracking surveys** and **swarming surveys.**

8.2 Bat activity surveys – manual and automated/static

8.2.1 Description and aims

Manual bat activity **transect surveys** involve ecologists walking predetermined transect routes in order to observe, listen for and record bats in flight away from their roosts using hand-held bat detectors and recorders. **Automated/static activity surveys** involve bat detectors being deployed at fixed locations to record bat activity remotely. These are usually used in combination with transect surveys.

The aim of these surveys is to answer the questions posed in Section 8.1. The results of these surveys can then be used to inform the need for further surveys or to facilitate an impact assessment and the subsequent design of appropriate mitigation.

8.2.2 Equipment

Generic documentation/equipment required for field surveys for bats is provided in Section 2.5.2; survey-specific equipment is listed in Appendix 1.

8.2.3 Expertise and licences

Section 2.5.1 discusses expertise and Section 1.2.2 provides information on licences. Activity surveys are unlikely to disturb bats if carried out correctly; however, it is good practice for these surveys to be designed and carried out, or at least led, by licensed surveyors who have gone through a period of training and evaluation.

8.2.4 Methods

8.2.4.1 Transect surveys

Appropriate transect routes should be determined during the fieldwork carried out as part of the preliminary ecological appraisal (see Chapter 4). This survey should have identified the different habitats in the survey area that will be impacted by the proposed activities and may have assessed suitability (see Section 4.3.4). All habitats should be sampled but the habitats identified as having moderate or high suitability for bats are likely to be the main focus of the transect surveys.

The extent and arrangement of the different habitats on site should inform the number and arrangement of transects required to complete the survey. This is also influenced by ease of accessibility and navigation. Some habitat types (for example, wetlands or dense scrub/woodlands) may constrain transect surveys and increase the emphasis on collecting data from spot counts, timed searches (see Section 8.2.5) or automated/static surveys (see Section 8.2.4.2).

Ideally, ecologists should have the opportunity to walk transects during the daytime in order to avoid getting lost; to identify hedge or watercourse crossing points; and to identify any particular hazards. It is more appropriate for this work to be carried out in pairs; for ecologists to know where other colleagues will be on site; and for the method of communication to be identified. This may require two-way radios in the absence of mobile phone signal.

During transect surveys, an ecologist should walk at a fairly constant speed (so the sampling area is the same per unit time) along a planned route recording observations of bats such as number of bats, flight direction, flight height, behaviour (e.g. commuting or foraging – the latter can be identified through hearing feeding buzzes), appearance and relative speed. Much of this is qualitative information that cannot be recorded using the automated systems described in Section 8.2.4.2, although obviously constrained by light levels (more so in cluttered habitats). All echolocation calls should be recorded and subsequently analysed to species or genus (see Chapter 10) even if the ecologist has attempted to identify the species by ear in the field.

Technology is available to record each bat echolocation call and link it to a specific location (using GPS points) and time to enable the data to be easily mapped and presented in reports, although some ecologists still use paper recording forms to record time, location, species and behaviour.

Because an ecologist is only in one location at a given time, it is likely that bat activity will be missed. Repeating a short transect twice during the course of one evening, randomly varying the starting point through the season and/or supplementing transect surveys with automated/static detector surveys can help to overcome this limitation. Different methods facilitate different types of analysis. For example, randomising the start point across a suite of surveys facilitates the production of a kernel density plot of the activity along the transect (see Figure A7.6).

Ideally, all habitats represented on site should be sampled by transects during a single survey visit to allow a comparison of bat activity across the site. However, if few ecologists are available and the site is particularly large it may be necessary to visit the site (covering different transects each time) over several consecutive nights (collectively considered to be 'one survey visit') to cover all areas.

Transect surveys can be undertaken as:

- **dusk surveys only** – this is likely to be the most effective method in the spring and autumn when conditions are likely to deteriorate in the night and may cause bats to go back to their roosts and not emerge for a second time before dawn;

- **dusk and pre-dawn surveys with a break between the two** – this is a useful method if the conditions are appropriate for pre-dawn activity but long nights mean a dip in bat activity is experienced in the middle of the night;

- **dusk to pre-dawn surveys** – this is most useful on short summer nights when activity levels remain high, or where the aim is to record particular types of bat activity in the middle of the night such as mating or swarming along with dusk and dawn activity;

- **pre-dawn surveys only** – these may be used to record specific pre-dawn behaviours such as bats commuting back to a roost in a particular direction.

Where multiple transects are carried out at one site, they should all be approximately the same length. A good guide is 3–5km, but transects may be shorter than this depending on the site, ground conditions, whether or not stopping points are used and levels of bat activity.

8.2.4.2 Static/automated surveys

The use of static/automated detectors facilitates quantitative analysis of the data to supplement the often qualitative data collected during transect surveys. Some examples of strategies that can be used to identify bat detector locations are given below (please refer back to Section 2.2.6 on data analysis):

- **Random:** a random sampling strategy is a good method for not introducing bias (distortion) to the subsequent analyses. The survey area is divided up into a grid (10 × 10) of equal squares and rather than surveying all squares, 20 squares are chosen randomly from the 100. Each square is numbered from 1 to 100 and 20 numbers are generated randomly, between 1 and 100, and assigned to a square.

- **Systematic:** a systematic sampling strategy is a good method for not introducing bias (distortion) to the subsequent analyses. The survey area is divided up into a grid (10 × 10) of equal squares and rather than surveying all squares, every 5th square is chosen. Each square is numbered sequentially 1 to 100 and then squares 1, 5, 10, 15, 20, etc... are selected.

- **Judgemental:** sampling locations in the survey area are chosen subjectively. For example 20 sampling locations, using the example area above, are determined based on expert opinion (after the preliminary ecological appraisal; see Chapter 4) or historical information. The approach could be described as 'haphazard' and at the extreme can fall into 'convenience' (sampling at convenient places (or times)). Judgement sampling has inherent uncertainty, cannot be readily quantified and statistical methods cannot be applied. However, this approach may facilitate the chances of recording, for example, quieter calling bats (see Table 3.7 on page 31).

- **Stratified:** the survey area is divided unequally into sub-areas allowing a sub-area(s) of interest to be surveyed more intensively (identified during the preliminary ecological appraisal; see Chapter 4). Sub-areas can be analysed individually but care should be taken when looking at the area as a whole because a bias has been introduced; some areas have been surveyed more than others. One way of looking at the whole area, while surveying sub-areas more intensively, is to pair or group sample locations by factors and use the factors in the analysis. Factors are most useful when they are simple and easily defined:
  - Field 1 – Field 2 (adjacent to Field 1 and same area)
  - Hedgerow – Watercourse (same length)
  - Woodland – Open field (same area)

Random, systematic, judgemental and stratified sampling strategies also apply to the timing of surveys; the convention in bat surveying is to use timings that are systematic.

Ideally, the same model of automated/static bat detector should be used across the site, all detectors should be deployed with the same settings and all detectors should be subject to regular testing/calibration as appropriate to avoid the introduction of bias and to allow a meaningful comparison of the results.

The microphone should be positioned to maximise the amount of bat activity recorded – this requires knowledge and consideration of the directionality/sensitivity of the particular microphone used. The choice of microphone (uni- or omnidirectional) will depend on the objectives of the survey – both types have their uses.

Automated/static detectors may be deployed at varying heights depending on site and project-specific factors. It is not usually appropriate to deploy a detector on the ground because this will decrease the survey volume around the microphone. The microphone should be located so that the recording of ambient (e.g. wind, running water, rustling vegetation) or any other source of extraneous noise (e.g. electrical signals) is minimised. It is also important to consider whether solid objects nearby (e.g. vegetation, built structures, etc.) will impede the passage of sound to the microphone, and adjust its position accordingly. It may be appropriate to elevate the detector above the height of a hedgerow to enable recording on both sides or to deploy the detector just below or above the canopy of a woodland. It may also be necessary to fence the detector or have livestock moved from a field if surveying in open pasture is necessary.

Data from automated/static systems is limited because there is no observational context. One hundred bat passes could represent one bat passing 100 times or 100 bats each passing once. Reality is likely to be somewhere between these two extremes. In cases where high levels of activity are recorded it may therefore be necessary to contextualise the results (i.e. is it one bat or 100 bats) using a manual transect or spot count survey. These methods are complementary – each performs a different function.

Table 8.1 provides a summary of the comparative benefits and limitations of transect and automated/static surveys.

Table 8.1 A summary of the comparative benefits and limitations of transect and automated/static surveys.

Survey type	Benefits	Limitations
Transect	• Bats can be counted • Bat behaviour can be observed (more limited as light falls and in cluttered habitats)	• Snapshot of time only • Ecologist is only in one location at any given time so could miss activity elsewhere • Subjectivity of ecologist can limit consistency, repeatability and quantitative analysis • Security of ecologists • Difficult in some habitat types (e.g. dense woodland or scrub or open homogenous habitats) • Labour-intensive fieldwork • Can't be used at height
Automated/static	• Can be deployed for long periods to pick up variability in bat activity in the absence of ecologists • Can be deployed in different locations simultaneously • Large amounts of data generated • More objective and therefore consistent, repeatable and allows quantitative analysis • Full auto identification is possible with some models, although caution should be exercised in choice and accuracy of software and reliance on results • Can be used very effectively for at-height surveys	• Bats cannot be counted • Bat behaviour cannot be observed • Large amounts of data generated, requiring significant storage capability • Lots of data analysis • Variability of weather over longer periods (though evens out over longer periods) • Security of detectors • Need to change memory cards and batteries

The results of the surveys should be recorded in a standard format and survey design should be iterative, each survey informed by the previous one. This is particularly important for automated surveys, where issues with a particular site or piece of equipment that would not otherwise be apparent may need to be addressed.

8.2.5 Complementary/alternative methods

Transect surveys may be supplemented by **spot counts**, where ecologists remain stationary for short, set periods of time (3–5 minutes) at locations along a transect route selected to represent the different habitats in the survey area. It may be appropriate to only sample at the spot count locations (rather than also recording along the transect) in habitats that are difficult to navigate or walk such as dense woodlands, wetlands or on steep terrain. This can make hearing, observing and counting bats easier because there is no noise from footfall and the ecologist can focus on the survey rather than navigation and safety.

Timed searches allow ecologists to move freely around the survey area for a set amount of time responding to any visual or acoustic evidence of bats by moving towards it. Timed searches

can be used to standardise survey methods for bat species that are difficult to detect, or if bats are spread over a wide area that cannot easily be sampled using transects or spot counts. Timed counts provide a simple and effective means of obtaining estimates of relative bat activity in homogeneous or difficult terrain such as mountains or wetland bogs; landscapes with few features (moor, open farmland); and areas where it is difficult to walk around (e.g. in dense woodland, built-up areas, railway marshalling yards, etc.). Large sites can be subdivided into smaller areas; a random sample of these can be selected for sampling or each can be sampled on a different night. Searching for a set amount of time introduces an element of standardisation that can be repeated in subsequent surveys.

Vantage point surveys can provide information about the behaviour of early-emerging and high-flying bats such as noctule. Ecologists are located at vantage points around the site, so that all areas are covered. They then observe and listen for bats in flight while light levels allow, before and after sunset or sunrise. These surveys can provide information about numbers of bats and direction of travel, which gives an indication of the direction of the roost and the direction of early evening foraging grounds.

Transect surveys have been carried out using bikes or cars to cover more ground (or boats in aquatic habitats). However, the limitations of these methods should be recognised. Car surveys are particularly constrained because they focus the survey only on roads/tracks and the noise and lights of the cars could disturb some bat species (particularly species that avoid light). Quieter-calling species can easily be missed so these methods should not be used in isolation.

It may be necessary to capture bats using mist nets or harp traps in order to identify their species, gender and age to supplement activity survey information. The correct licence, skills and experience should be in place to carry out this activity and sensitive times of year should be avoided (such as the maternity and hibernation seasons). More information on capture and handling is provided in Chapter 9.

8.2.6 Timing

Recorded bat activity is dependent on the prevailing conditions at the time of the survey, which vary temporally (through the night, between nights, through the seasons and between years) and spatially (dependent on latitude, longitude, altitude, habitat, etc.).

Bat activity is also determined by what the bats are doing at different times of the year (although this is also dependent on prevailing conditions); the bat life cycle is given in Section 3.2.

The UK bat active period is generally considered to be between April and October inclusive, although April, September and October surveys are both weather- and location-dependent (October surveys are generally not acceptable in Scotland). Conditions may become more unsuitable in these months, particularly in more northerly latitudes, which may reduce the length of the survey season. Some useful data may be collected outside these months or weather conditions during these months may render surveys ineffective – professional judgement should be applied to determine the most effective activity survey period for a particular project.

It may be appropriate to survey for bat activity in the winter, particularly if there are hibernation roosts in, or close to, the survey area. Foraging habitats close to hibernacula may be particularly important because during the winter bats need to minimise energy used to gain food during milder weather conditions. Automated/static surveys are likely to be the most efficient way of collecting data on winter bat activity.

Table 8.2 gives recommended timings for activity surveys.

Table 8.2 Recommended start and end times for activity surveys.

Survey type	Start time	End time
Dusk survey – bat activity	Sunset[a]	2–3 hours after sunset
Dusk survey – swarming	2 hours after sunset	5 hours after sunset
Dusk to pre-dawn survey	Sunset	Sunrise or later if bats still active
Pre-dawn survey	2 hours before sunrise	Sunrise or later if bats still active
Automated bat detector survey	30 minutes before sunset	30 minutes after sunrise

[a] Adjust to earlier if in darker habitats such as woodland or if data justifies (e.g. if bats are already out by sunset on previous surveys or automated detectors show pre-sunset activity).

Timings may be adjusted (earlier or later) if necessary by the ecologist, bearing in mind the site-specific circumstances, although this should be fully justified in the survey report.

8.2.7 Survey effort

When planning surveys it is important to take a proportional approach. The number of transects, automated/static surveys and repeat visits decided upon should be proportional to the factors described in Section 2.2.5. To briefly recap, with particular reference to activity surveys, these are:

- likelihood of bats being present;
- likely species concerned;
- numbers of individuals;
- type of habitat affected;
- predicted impacts of the proposed development on bats;
- type and scale of proposed development.

An activity survey should provide a representative sample of the bat activity in all habitats present at the proposed development site (see Section 8.2.4.1). Sampling should be designed to provide a sufficient amount of data to assess the potential impacts of the development on bats.

Bat activity is inherently variable from night to night, with this variability not explained by weather conditions alone (Scott and Altringham, 2014), and so multiple consecutive nights of survey with automated systems are recommended.

Table 8.3 gives guidelines on the number of bat activity surveys recommended to achieve a reasonable survey effort on sites with low, moderate and high-quality habitat for bats (as defined during the preliminary ecological appraisal fieldwork; see Table 4.1 on page 35). Please note that the elements outlined in Section 2.2 should be considered alongside habitat suitability in designing surveys. In particular, the potential impacts of the proposals (Section 2.2.2) and proportionality (Section 2.2.5).

Table 8.3 Guidelines on the number of bat activity surveys recommended to achieve a reasonable survey effort in relation to habitat suitability.

Survey type	Low suitability habitat for bats[a]	Moderate suitability habitat for bats	High suitability habitat for bats
Transect/spot count/timed search surveys	One survey visit[b] per season (spring – April/May, summer – June/July/August, autumn – September/October)[c] in appropriate weather conditions for bats Further surveys may be required if these survey visits reveal higher levels of bat activity than predicted by habitat alone	One survey visit[b] per month (April to October)[c] in appropriate weather conditions for bats. At least one of the surveys should comprise dusk and pre-dawn (or dusk to dawn) within one 24-hour period.	Up to two survey visits[b] per month (April to October)[c] in appropriate weather conditions for bats. At least one of the surveys should comprise dusk and pre-dawn (or dusk to dawn) within one 24-hour period.
AND			
Automated/static bat detector surveys[d]	One location per transect, data to be collected on five consecutive nights per season (spring – April/May, summer – June/July/August, autumn – September/October)[c] in appropriate weather conditions for bats	Two locations per transect, data to be collected on five consecutive nights per month (April to October)[c] in appropriate weather conditions for bats	Three locations per transect, data to be collected on five consecutive nights per month (April to October)[c] in appropriate weather conditions for bats

[a] If the habitat has been classified as having low suitability for bats, an ecologist should make a professional judgement on how to proceed based on all of the evidence available. It may or may not be appropriate for bat activity surveys to be carried out in low suitability habitats. However, caution should be exercised in fringe areas (e.g. some areas of Scotland) where 'low suitability habitat for bats' may be extremely important to local bat populations due to the relative scarcity of better habitats. In such situations, bats are likely to also be more widely dispersed and may use a larger number of sites, therefore survey effort may actually need to be increased to detect use on the proposed site in question.

[b] A survey visit should aim to cover all habitats represented in the survey area that could be impacted by the proposed activities. This may consist of a single transect carried out on a single night for small sites (e.g. small housing developments) with low habitat diversity but could range up to multiple transects carried out over one or several nights (depending on number of ecologists) on a larger site (e.g. road schemes) with greater habitat diversity.

[c] April, September and October surveys are both weather- and location-dependent. Conditions may become more unsuitable in these months, particularly in Scotland, which may reduce the length of the survey season.

[d] Detector locations should be assigned to cover all habitats represented in the survey area that could be impacted by the proposed activities. This could mean a single detector location at a small site with only one habitat represented but could range up to many detector locations on larger sites. Automated/static surveys are particularly useful when assessing collision risk, e.g. detectors can be placed at crossing points on proposed roads or railways.

Note: Multiple survey visits should be separated by at least two weeks, preferably longer, to observe temporal changes in activity.

Survey data should be analysed as soon as possible and preferably before the next survey (see Section 10.2).

It is important to consider how effective the surveys are in recording species that are more difficult to detect (see Section 3.9) or exhibit highly variable or seasonal patterns of activity. It may be appropriate to adjust the survey methods, increase the number of survey nights or adjust the survey frequency to ensure these species are not under-recorded. Skalak *et al*. (2012) reported that relatively few nights are needed to detect common species but longer sampling periods may be necessary to detect rarer species. The same is true of those species that use quiet echolocation calls (see Table 3.8 on page 32).

Comparing transect and static data may also indicate that species are being recorded by one type of survey but not another, so that subsequent surveys can be adjusted accordingly.

8.2.8 Weather conditions
Please refer to Section 2.6.1 for guidance on weather.

8.2.9 Next steps
The next steps will depend on what has been recorded during the activity surveys. It may be necessary to carry out more targeted activity surveys in subsequent years or use alternative methods to gain specific information (e.g. using a trapping survey to distinguish between *Myotis* or *Plecotus* species or to define breeding status of the bats; see Chapter 9).

Where enough information has been collected, the data should be used to inform an impact assessment and the design of a mitigation strategy.

8.3 Swarming surveys – acoustic

8.3.1 Description and aims
Swarming surveys are carried out to identify if a site is used by bats for autumn swarming, which was described by Fenton (1970) as 'the flight of bats through hibernacula in late summer and early fall'. This usually occurs in the UK from August to October inclusive and activity peaks 3–4 hours after sunset (Rivers *et al.*, 2006; Glover and Altringham, 2008): observations made during the first 2–3 hours after sunset may not detect it. Autumn swarming should not be confused with what is commonly termed 'dawn swarming', where one or more bats fly around outside their roosts prior to entry at dawn.

Autumn swarming behaviour has been recorded mostly at the entrances to and outside underground sites such as caves, mines and tunnels but has also been observed around other structures such as castles, and large barns. Evidence from the Netherlands shows mass swarming events of common pipistrelle bats in the autumn followed by mass hibernation in a diverse range of building types in urban environments (Korsten *et al*., 2015). This phenomenon requires some research in the UK but ecologists should be aware of the potential for larger numbers of this species to be present during the autumn and winter in large buildings in highly urbanised environments.

Swarming behaviour is common among *Myotis*, *Plecotus* and *Barbastella* species. Swarming probably has several important functions: mating, transfer of information about hibernation sites to young, collection of information on the condition of hibernation sites prior to hibernation and migration stopover, but as yet most lack direct evidence to support them. There is, however, good behavioural and genetic evidence to show that mating is an important function (Thomas *et al.*, 1979; Kerth *et al.*, 2003; Rivers *et al.*, 2005; Furmankiewicz and Altringham, 2007).

Rivers *et al.* (2006), in a study of four North Yorkshire caves, found that Natterer's bats undertook seasonal migration between the caves and their nursery sites over at least a 60km radius area. Between 300 and 400 bats visited the caves each night, with many more at the peak of the season. Numbers of bats vary between sites and from night to night at the same site. Activity typically starts in August and rises to a peak in September or early October before slowly declining. Many thousands of bats may visit some sites, but swarming behaviour may involve no more than a few bats each night at minor sites.

Swarming sites can therefore be important mating sites for large numbers of bats and are important for gene flow (Kerth *et al.*, 2003; Rivers *et al.*, 2005; Furmankiewicz and Altringham, 2007). Many underground swarming sites are also hibernation sites and it is likely that those bats swarming at a site go on to hibernate in the same site (Glover and Altringham, 2008). Individual bats show very high fidelity to a single swarming site (Rivers *et al.*, 2005, 2006; Glover and Altringham, 2008) and few bats are recaptured at other sites, even those close by.

The impact of destroying or changing a swarming site for development purposes is likely to be severe, so it is particularly important to investigate further whether swarming is a possibility. The aim of carrying out acoustic bat activity surveys at potential swarming sites is to establish actual use of the site by swarming bats and understand how bats use the site. If a site likely to be impacted by development does support swarming bats it is possible that further surveys will be necessary (see Chapter 9).

8.3.2 Equipment
Generic documentation/equipment required for field surveys for bats is provided in Section 2.5.2; survey-specific equipment is listed in Appendix 1.

8.3.3 Expertise and licences
Section 2.5.1 discusses expertise and Section 1.2.2 provides information on licences. Acoustic swarming surveys are unlikely to disturb bats if carried out correctly; however, it is good practice for these surveys to be designed and carried out, or at least led, by licensed surveyors who have gone through a period of training and evaluation. If disturbance to bats is likely (e.g. because an ecologist needs to deploy a detector in the entrance to a roost), then a survey licence is required.

8.3.4 Methods
The simplest and most efficient way to investigate whether bats are swarming at a site is to deploy an automated/static bat detector and recorder to record swarming bats just outside or within the entrance to an underground site (or complex structure). Repeated peaks in ultrasonic activity, reaching a maximum 3–4 hours after sunset, indicate the site is used by swarming bats and the echolocation calls recorded can be analysed to species or genus after the survey. This method is likely to generate a large amount of data at a swarming site because of the high levels of activity generally observed. However, it is unlikely to be necessary to scrutinise all recordings made, depending on the aims and objectives of the survey. Alternatively, data collected could be reduced by recording for only a few hours during the middle of the night (e.g. 2–5 hours after sunset).

8.3.5 Complementary methods
It may be appropriate to trap bats at a swarming site if it is necessary to confirm species, particularly if Annex II species such as barbastelle and Bechstein's bat may be present. Wherever and whenever possible, harp traps should be used in preference to mist nets due to the possibility of catching large numbers of bats. More information on trapping is provided in Chapter 9. With the evolution of more reliable software for automated identification from echolocation calls, trapping to determine species only may eventually become unnecessary.

Trapping to establish gender is unnecessary because the pattern of use at swarming sites is well documented: both sexes are present, but males outnumber females, consistent with mating behaviour during swarming (Thomas *et al.*, 1979; Kerth *et al.*, 2003; Parsons *et al.*, 2003a; Rivers *et al.*, 2005; Furmankiewicz and Altringham, 2007; Glover and Altringham, 2008).

8.3.6 Timing

Swarming surveys should be carried out from mid-August to October inclusive, but if only a limited survey period is available mid-September to early October is best. Species composition varies throughout the swarming season, with *Plecotus* and most *Myotis* species either peaking early or showing no discernible peak, and Natterer's bat peaking late in the season (Parsons *et al.*, 2003b; Rivers *et al.*, 2006; Glover and Altringham, 2008). Bechstein's bat, Alcathoe bat and barbastelle also swarm, but numbers caught are too low to reveal temporal patterns. Most sites have a similar mix of species: brown long-eared bat and the *Myotis* species expected in the area. It is typical for Natterer's bat to greatly outnumber all other species, particularly from mid-season onwards. Non-swarming species may also be recorded, particularly horseshoe bats, depending upon the nature of the adjacent habitat.

8.3.7 Survey effort

At least five nights of survey with an automated/static detector (in appropriate weather conditions for bats; see Section 2.6.1) in each month of the swarming season of mid-August to the end of October is recommended to establish whether a site is used for swarming or not.

If trapping is undertaken, then recommendations on survey effort are provided in Chapter 9.

8.3.8 Weather conditions

Please refer to Section 2.6.1 for guidance on weather.

Many studies have noted that bat activity at swarming sites varies markedly from night to night: bat activity is significantly suppressed by rainfall and positively correlated with residual maximum ambient temperature. Grubb (2012) also found high winds depressed activity. Moon phase does not appear to influence swarming activity (Parsons *et al.*, 2003a), but a bright moon has been known to lower capture success (if trapping) at exposed locations. Swarming activity appears to be more likely when weather conditions are more stable so targeting periods of high pressure may be appropriate.

8.3.9 Next steps

See Chapter 9 regarding trapping bats at swarming sites. This is only likely to be necessary if Annex II species may be present. If the presence of Annex II species is unlikely, then trapping is less appropriate because species assemblages using swarming sites are well documented from other studies (Parsons *et al.*, 2003b; Rivers *et al.*, 2006; Glover and Altringham, 2008).

Swarming sites are also used for hibernation so it may be necessary to also carry out hibernation surveys as described in Section 5.3.

8.4 Back-tracking surveys

8.4.1 Description and aims

Back-tracking surveys involve ecologists making visual observations of bats commuting away from their roosts at sunset or commuting back to their roosts at sunrise then attempting to track back to the roost based on these observations. Bat detectors are also used to record echolocation for identification of species, where possible. This technique was first developed in the Netherlands and is based on four principles:

- The earlier a bat is seen after sunset or the later it is seen before sunrise, the closer it is likely to be to its roost (the exact time depends on the species).
- Bats fly away from their roost at sunset, so ecologists should move in the opposite direction as the bats at this time to locate the roost.
- Bats fly towards their roost at sunrise, so ecologists should move in the same direction as the bats at this time to locate the roost.
- At sunrise, some bats species swarm at roost access points for between 10 and 90 minutes before entering.

The aim is to find roosts by making observations of commuting bats. These surveys are often used after a bat activity survey if numbers of bats were seen all commuting in one direction and follow-up is required or in situations with lots of potential roosts sites that are difficult to survey using alternative methods (e.g. in woodlands or highly urbanised areas).

8.4.2 Equipment

Generic documentation/equipment required for field surveys for bats is provided in Section 2.5.2; survey-specific equipment is listed in Appendix 1.

8.4.3 Expertise and licences

Section 2.5.1 discusses expertise and Section 1.2.2 provides information on licences. Back-tracking surveys are unlikely to disturb bats if carried out correctly; however, it is good practice for these surveys to be designed and carried out, or at least led, by licensed surveyors who have gone through a period of training and evaluation. If disturbance to bats is likely (e.g. because an ecologist would needs to investigate the roost when found using a torch or endoscope), then a survey licence would be required.

8.4.4 Methods

Ecologists should be deployed on potential or actual commuting routes close to roost sources (identified during the preliminary ecological appraisal, see Chapter 4, or during activity surveys, see Section 8.2) and note the time and direction of travel of each bat encountered on a detailed plan of the site. The ecologists should move in the opposite direction to the bats at sunset and in the same direction as the bats at sunrise. As ecologists approach potential roosts they should watch for bats emerging or dawn swarming at roosts.

If multiple ecologists are used they should be in constant contact via hand-held radio to communicate their observations. The data from multiple ecologists can also be pooled for a bigger picture of bat activity across the site, which can be used to design subsequent surveys where necessary.

In theory, back-tracking surveys work best for species with loud echolocation calls which form large roosts, but they can potentially be used to locate the roosts of any bat species.

8.4.5 Complementary methods

Back-tracking surveys are rarely used in isolation; they are most effective when combined with roost (Chapters 5 to 7) and bat activity surveys (Section 8.2).

8.4.6 Timing

As back-tracking surveys are most effective for larger roosts, the best time to carry them out is between May and August, when maternity colonies are gathered. However, results may be gained if carried out in April, September or October, depending on the individual situation (although October surveys are not considered appropriate in Scotland). See Section 8.2.6 for further comments on timing of activity surveys through the year. Table 8.4 gives recommended timings for back-tracking surveys during the night.

Table 8.4 Recommended start and end times for back-tracking surveys.

Survey type	Start time	End time
Back-tracking survey at dusk	15 minutes before sunset	When it is too dark to observe bats or when the source roost has been found
Back-tracking survey at dawn	2 hours before sunrise	When bats cease to be active or when the source roost has been found

Timings may be adjusted (earlier or later) if necessary by the ecologist, bearing in mind the site-specific circumstances, although this should be justified in the survey report.

8.4.7 Survey effort

The survey effort for back-tracking surveys is not fixed. These surveys have the specific aim of locating roosts using commuting bats for guidance and should be continued until this aim is reached unless alternative methods are considered more appropriate.

8.4.8 Weather conditions

Please refer to Section 2.6.1 for guidance on weather.

8.4.9 Next steps

If a roost is found during a back-tracking survey it may be necessary to follow up with a roost characterisation survey (see Section 7.2) to count the numbers of bats present at the roost.

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COUNCIL OF EUROPE
COMMITTEE OF MINISTERS

RESOLUTION (79) 13

ON THE AWARD OF THE EUROPEAN DIPLOMA TO THE MINSMERE NATURE RESERVE (UNITED KINGDOM)

(Adopted by the Committee of Ministers on 13 September 1979 at the 307th meeting of the Ministers' Deputies)

The Committee of Ministers, under the terms of Article 15.*a* of the Statute of the Council of Europe,

Having regard to Resolution (65) 6 instituting the European Diploma ;

Having regard to the proposals of the European Committee for the Conservation of Nature and Natural Resources ;

Having recorded the agreement of the Government of the United Kingdom ;

After deliberation,

Solemnly awards the European Diploma, Category A, in accordance with the regulations for the European Diploma, to the Minsmere Nature Reserve (United Kingdom) ;

Places the aforesaid reserve under the patronage of the Council of Europe until 13 September 1984.

COMMITTEE OF MINISTERS
COMITÉ DES MINISTRES
1949.2019
COUNCIL OF EUROPE
CONSEIL DE L'EUROPE

MINISTERS' DEPUTIES Resolutions CM/ResDip(2019)5 4 September 2019

Resolution CM/ResDip(2019)5 on the renewal of the European Diploma for Protected Areas awarded to the Minsmere Nature Reserve (United Kingdom)

(Adopted by the Committee of Ministers on 4 September 2019 at the 1352nd meeting of the Ministers' Deputies)

The Committee of Ministers, under the terms of Article 15.*a* of the Statute of the Council of Europe,

Having regard to Resolution Res(65)6 instituting the European Diploma for certain protected landscapes, reserves and natural features, as amended by Resolution CM/ResDip(2008)1 on the revised regulations for the European Diploma for Protected Areas;

Having regard to Resolution Res(79)13 on the award of the European Diploma to the Minsmere Nature Reserve (United Kingdom);

Having regard to Resolution CM/ResDip(2009)9 on the renewal of the European Diploma for Protected Areas awarded to the Minsmere Nature Reserve (United Kingdom);

Taking into consideration the expert's report presented at the meeting of the Group of Specialists on the European Diploma for Protected Areas on 5 and 6 March 2019;

Having regard to the proposals of the Standing Committee to the Bern Convention;

Noting that holding the European Diploma implies the maintenance of exceptional European interest, the continuance of exemplary management and compliance with the provisions of Resolution CM/ResDip(2008)1, especially with regard to the obligations to report annually and to give appropriate visibility to the European Diploma;

Renews the European Diploma for Protected Areas awarded to the Minsmere Nature Reserve, until 13 September 2029;

Attaches the following condition to the renewal:

with regard to the project for a third reactor at the Sizewell nuclear power plant (project Sizewell C), carry out a proper Strategic Environmental Assessment and an Environmental Impact Assessment according to international standards; the results of these assessments should ensure that the construction of the new reactor will not be to the detriment of the Minsmere Nature Reserve;

Attaches the following four recommendations to the renewal:

1. control, wherever possible, the invasion of bracken with mechanical means, as opposed to the use of herbicides; in cases of restoration of big areas, the use of chemical products should be limited to the minimum possible extent and a careful record of this practice should be kept in order to assess and cope with possible undesirable side effects; specific scientific research on this issue should be carried out;

2. pursue efforts to mitigate the effect of the rise in the sea level; explore the possibility of restoring the lost habitats in other parts of the reserve; envisage compensatory measures; carefully monitor indicative plant and fauna species and duly register and scientifically characterise the changes observed;

3. continue the land purchase policy; in the absence of opportunities to acquire new plots, establish long-term agreements with the land owners of selected areas to improve their contribution to biodiversity conservation and/or restoration;

4. maintain the surveillance of alien species of flora, especially controlling and managing the pirri-pirri bur (*Acaena novae-zelandiae*) and pursue the control of the American mink (*Mustela vison*).

Natural England Joint Publication JP029

The Biodiversity Metric 2.0

auditing and accounting for biodiversity

USER GUIDE

Beta Version

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1: Introduction

The rationale for using a metric

1.1. Biodiversity is the term that is used to describe the variety of all life on earth. It includes all species of animals and plants – and everything else that is alive on our planet. Habitats are the places in which species live. These species and their habitats contribute to the ecosystems services that provide substantial benefits to people and the economy. For example, woodlands and saltmarsh can help prevent flooding whilst parks and greenspaces make our towns and cities healthier and more attractive places in which to live and work. However, biodiversity is under threat, globally and at home. Habitats are being damaged or disappearing and species are declining. This is not just bad news for nature but also for our own health and well-being and that of future generations. Biodiversity and healthy habitats are vital for a well-functioning planet but their value is often not taken into account in decision-making.

1.2. In this user guide we introduce and explain how to use the **biodiversity metric 2.0**. This metric provides a way to measure biodiversity and the impact that developments or land management practices may have upon it. Biodiversity metric 2.0 can help developers, ecologists, planners, communities, land managers and many others take biodiversity into account. The metric provides a way to measure biodiversity loss and gain in a consistent and robust way. It can also predict the likely effectiveness of creating new or enhancing existing habitats. Used in combination with appropriate professional advice the metric can help to reduce biodiversity losses and increase gains resulting from development or land management.

Introducing the biodiversity metric 2.0

1.3. Biodiversity metric 2.0 is an updated version of the original Defra biodiversity metric[1]. This version builds upon the knowledge and experience gained across a variety of different sectors since the original Defra biodiversity metric was first launched as part of Defra's biodiversity offsetting pilots.

1.4. Biodiversity metric 2.0 balances robustness with simplicity. The metric uses habitat as a proxy for wider biodiversity with different habitat types scored according to their relative biodiversity value. This value is then adjusted depending on the condition and location of the habitat, to calculate '**biodiversity units**' for that specific project or development. Biodiversity metric 2.0 incorporates similar but separate calculations for habitats that require a different method of measurement such as hedgerows, lines of trees, rivers and streams and street trees.

1.5. The metric can be used to measure both on-site and off-site biodiversity changes for a project or development. The metric also accounts within it for some of the risks associated whenever new habitat is created or existing habitat is enhanced. In calculation terms, the change in biodiversity units is determined by subtracting the number of **pre-intervention** biodiversity units (i.e. those originally existing on-site and off-site) from the number of **post-intervention** units (i.e. those projected to be provided). It is important to note that achieving gains in biodiversity from the

[1] DEFRA. 2012. Biodiversity offsetting pilots. Technical paper: the metric for the biodiversity offsetting pilot in England. Defra. March 2012. https://www.gov.uk/government/collections/biodiversity-offsetting (Accessed 20-06-2019)

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calculation does not necessarily mean a development meets any wider requirements of planning policy or law relating to nature conservation or biodiversity.

1.6. All biodiversity unit calculations come with some 'health warnings'. The outputs of the metric are not absolute values but provide a proxy for the relative biodiversity worth of a site pre- and post-intervention. The quality and reliability of outputs will depend on the quality of the inputs. This user guide provides advice on how to use the biodiversity unit approach and where and when it is appropriate for use. The metric is not a substitute for expert ecological advice. The metric does not override or undermine any existing planning policy or legislation, including the mitigation hierarchy (see section 1.11 below), which should always be considered as the metric is applied.

1.7. Biodiversity metric 2.0 does not include species explicitly. Instead, biodiversity metric 2.0 uses broad habitat categories as a proxy for the biodiversity 'value' of the species communities that make up different habitats. The metric does not change existing levels of species protection and the processes linked to protection regimes are outside the scope of the metric.

1.8. To simplify and streamline the calculation process, the biodiversity metric 2.0 comes with a free tool to calculate biodiversity units. A shortened user guide for the calculation tool is also available.

The mitigation hierarchy and the metric

1.9. Planning policy[2][3] supports the application of the **mitigation hierarchy** (see Figure 1-1). When using the metric application of the mitigation hierarchy might mean looking to retain habitats in situ or avoiding habitat damage. In the metric biodiversity gains are easier to achieve where habitat impacts are avoided due to the way that habitat creation or enhancement risks are accounted for.

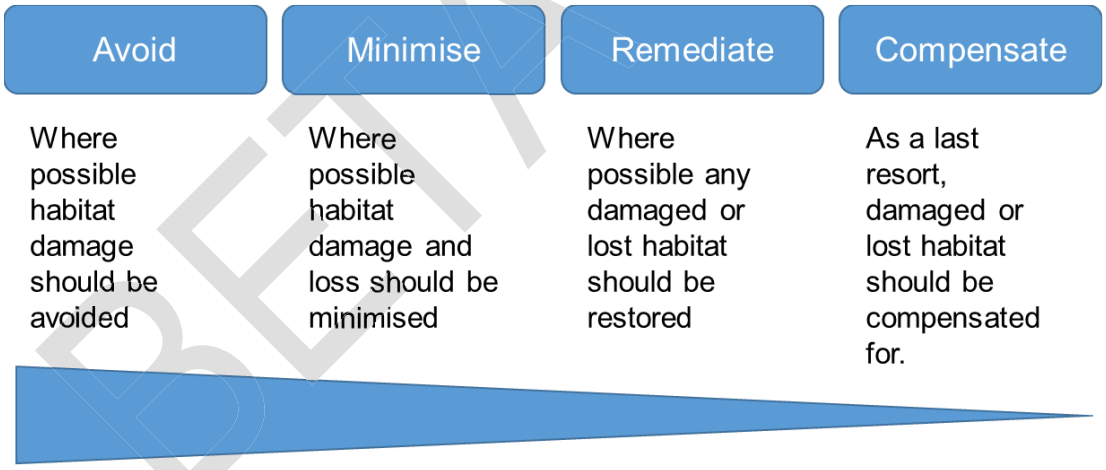


FIGURE 1-1: The Mitigation Hierarchy[4]

[2] Planning policy explained: https://www.gov.uk/guidance/national-planning-policy-framework
[3] NPPF implementation explained https://www.gov.uk/guidance/natural-environment
[4] Source: adapted from DEFRA, 2018, Net Gain Consultation Proposals. Defra, December 2018. https://consult.defra.gov.uk/land-use/net-gain/supporting_documents/netgainconsultationdocument.pdf (Accessed 20-06-2019)

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Key process steps

2.19. The key steps you need to follow to make practical use of the metric are outlined in Figure 2-1.

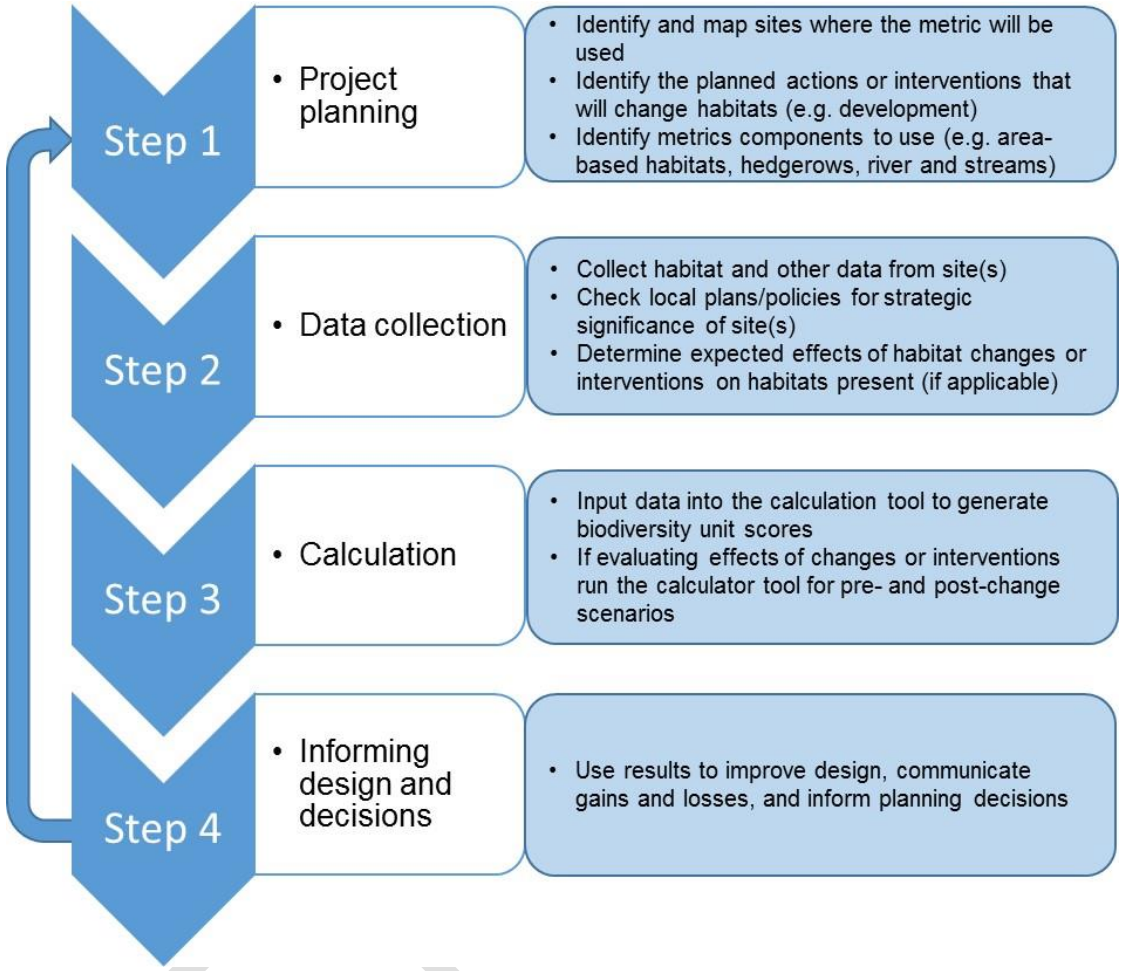


FIGURE 2-1: The 4 key steps to using biodiversity 2.0

Principles and rules for using the metric

2.20. The metric is a tool that can be used to help inform plans and decisions. Used properly, it incentivises actions that are expected to benefit biodiversity and discourages actions that harm biodiversity. It is important, however, to be aware of its limitations and to follow some important principles.

2.21. The metric uses habitat categories as a proxy for biodiversity. Although this is rational, it is an oversimplification of the real world. Furthermore, while the scoring of habitats is informed by ecological reasoning and the available evidence, the outputs of biodiversity unit calculations are not scientifically precise or absolute values. The generated biodiversity unit scores are proxies for the relative biodiversity worth for the state of a place.

2.22. The metric and its outputs should therefore be interpreted, alongside ecological expertise and common sense, as an element of the evidence that informs plans and

decisions. The metric is not a total solution to biodiversity decisions. The metric, for example, helps you work out how much new or restored habitat is needed to compensate for a loss of habitat, but it does not tell you the appropriate composition of plant species to use.

2.23. Users wanting to apply the metric properly should conduct their assessments with regard to a set of **key principles and rules** for its use. These are set out below:

- **Principle 1: The metric does not change the protection afforded to biodiversity.** Existing levels of protection afforded to protected species and to habitats are not changed by use of this or any other metric. Statutory obligations will still need to be satisfied.

- **Principle 2: Biodiversity metric calculations can inform decision-making where application of the mitigation hierarchy and good practice principles[6] conclude that compensation for habitat losses is justified.**

- **Principle 3: The metric's biodiversity units are only a proxy for biodiversity.** While it is underpinned by ecological evidence the metric is only a proxy for biodiversity and to be of practical use has been kept deliberately simple. The numerical values generated by the metric represent relative, not absolute, values.

- **Principle 4: The metric focuses on widespread species and typical habitats.** Area based habitats are considered a suitable proxy for widespread species found in typical examples of different habitat types.
    - Protected and locally important species needs are not considered through the metric,
    - Impacts on protected (e.g. SSSIs) and irreplaceable habitats are not adequately measured by this metric, and will likely require separate consideration.

- **Principle 5: The metric design aims to encourage enhancement, not transformation, of the natural environment.** Where possible, habitat created to compensate for loss of a natural or semi-natural habitat should be of the same broad type (e.g. new woodland to replace lost woodland) unless there is a good ecological reason to do otherwise (e.g. to restore a heathland habitat that was converted to woodland for timber in the past).

- **Principle 6: The metric is designed to inform decisions**. Decisions and management interventions need to take account of available expert ecological advice and not just the biodiversity unit outputs of the metric.

- **Principle 7: Compensation habitats should seek, where practical, to be local to the impact.** They should aim to replicate the characteristics of the habitats that have been lost, taking account of the structure and species composition that give habitats their local distinctiveness. Where possible

[6] CIEEM, CIRIA, IEMA. 2016 Biodiversity Net Gain – Good Practice Principles for Development. https://www.cieem.net/data/files/Publications/Biodiversity_Net_Gain_Principles.pdf

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compensation habitats should contribute to England's ecological network by creating more, bigger, better and joined areas for biodiversity

- **Principle 8: The metric does not enforce a mandatory minimum 1:1 habitat size ratio for losses and compensation.** A difference can occur because of a difference in quality between the site impacted and the compensation provided. For example, if a habitat of low distinctiveness is impacted and is compensated for by the creation of habitat of high distinctiveness, the area needed to compensate for losses can potentially be less than the area impacted. Consideration should be given to whether reducing the size of compensation is an appropriate outcome.

Rules	
Rule 1	Where the metric is used to measure change biodiversity unit values need to be calculated prior to the intervention and post-intervention for all parcels of land / linear features affected.
Rule 2	Compensation for habitat losses can be provided by creating new habitat, by restoring or enhancing existing habitats, or by accelerating successional processes. Measures to improve existing habitats must provide a significant and demonstrable uplift in distinctiveness and/or condition to record additional biodiversity units.
Rule 3	'Trading down' must be avoided. Losses of habitat are to be compensated for on a "like for like" or "like for better" basis. Ideally, new or restored habitats should aim to achieve a higher distinctiveness and / or condition than habitats lost.
Rule 4	Biodiversity unit values generated by biodiversity metric 2.0 are unique to this metric and cannot be compared to unit outputs from the original Defra metric or any other biodiversity metric. Furthermore, the units generated by the each module of biodiversity metric 2.0 (for area, hedgerow and river habitats) are unique and cannot be summed.
Rule 5	It is not the area of habitat created that determines whether ecological equivalence or better has been achieved but the net change in biodiversity units. Risks associated with enhancing or creating habitats mean that it may be necessary to enhance or create a larger area of habitat than lost to fully compensate for impacts on biodiversity.
Rule 6	Deviations from the published methodology of biodiversity metric 2.0 need to be ecologically justified. While the methodology is expected to be suitable in the majority of circumstances it is recognised that there may be exceptions. Any local or project-specific adaptations of the metric must be transparent and fully justified.

5: Detailed description of the biodiversity metric 2.0

5.1. This chapter provides explanations of the different components of biodiversity metric 2.0. The aim is to give a sense of the values used for different multipliers, why those multipliers are being used, and the assumptions and limitations around them. This chapter also outlines some the considerations that might be taken into account when designing a project underpinned by the metric.

5.2. This chapter focuses on the core components of the metric. Specific area habitat and urban tree components are detailed in chapters 6 and 7 respectively. Chapter 8 provides similar details and explanations for the additional supplementary metric modules covering hedgerows, lines of trees and rivers and streams.

Components of biodiversity quality

Distinctiveness

5.3. Habitats are assigned to distinctiveness bands. These are based on an assessment of the distinguishing features of a habitat or linear feature, including the consideration of species richness, rarity (at local, regional, national and international scales), and the degree to which a habitat supports species rarely found in other habitats.

5.4. The distinctiveness band of each habitat is preassigned in biodiversity metric 2.0. The bands are based upon the UK habitat classification system. A combination of simple rules and expert judgement have been used to assign each habitat type to the appropriate distinctiveness band. The distinctiveness categories used are tailored to habitat type and are explained later in this chapter for Area Habitats and in chapter 8 for habitats with supplementary modules.

Condition

5.5. Parcels of habitat will be in different ecological conditions In addition, interventions to improve habitats will not always involve taking a habitat in poor condition and improving it to good condition. The metric therefore takes account of variants in habitat condition.

5.6. The approach to condition assessment is tailored to habitat type and is explained later in this chapter for Area Habitats and in chapter 8 for habitats with supplementary modules.

Strategic significance and connectivity

5.7. ‘The spatial location of a habitat is treated as a component of the quality of a habitat parcel in the same way as distinctiveness or condition. Two distinct spatial components are used strategic significance and connectivity. These are explained in more detail in section 5.29.

Dealing with risk

5.8. There are uncertainties and a risk of failure in any endeavour to create or improve the biodiversity unit value of a habitat. One way to deal with these risks is to complete the habitat improvements works in advance of the habitat losses occurring.

5.9. Where this is not possible risks can be mitigated by reducing the number of units generated by a unit of compensation habitat. This is done by using a **multiplier** in the metric to correct for disparity or risk. The use of multipliers to account for the risks associated with habitat restoration or creation has several benefits:

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- it **provides flexibility** by allowing activities impacting habitats to proceed in advance of compensation being either provided or attaining its target quality in exchange for an increase in the magnitude of compensation provided;
- it **incentivises** the creation of compensation habitat in advance of loss. If the habitat is established before the impact then there is no need to apply risk multipliers to manage delivery risks or to take account of time differences. More units will therefore be available from a specific parcel of land, and
- it **creates a disincentive** for damaging habitats that are difficult or take a long time to recreate or restore (the case for many habitats in the Very High and High distinctiveness band), by increasing the area of habitat needed to compensate for the loss.

5.10. A typical consequence of applying risk multipliers is to increase the size (e.g. area or length for linear features) of habitat required as compensation such that it exceeds the size of habitat lost or damaged. This is necessary:

- to preserve the incentives and disincentives referred to above;
- to compensate for temporal losses of biodiversity (e.g. where there is a period of diminished biodiversity between the point in time when a habitat is impacted and it is replaced by habitat of equivalent biodiversity value);
- to protect against situations where habitats that are created, enhanced or restored fail to adequately compensate for the lost biodiversity. This is necessary because there is no requirement to provide additional compensation if interventions ultimately fail to deliver the predicted biodiversity outcome.

5.11. The following three risks are recognised in this metric.

Difficulty of creation and restoration

5.12. This risk associated with delivery of biodiversity creation or enhancement due to uncertainty in the effectiveness of management techniques used to restore or create habitat.

5.13. The level of risk differs between habitat types because of ecological factors (e.g. the different challenges posed by creating different habitat types) and due to the availability of techniques or know-how to create habitats in a realistic time-frame. Uncertainty in achieving the target outcome for each habitat is addressed by a habitat-specific 'difficulty' multiplier based on available science and expert opinion.

5.14. There is a growing body of experience and expertise associated with habitat creation and enhancement[11]. Nevertheless, it is important to recognise that it is impossible to exactly replicate habitat losses because of the unique physical and ecological features of every place. This point is of particular relevance to impacts on well-established semi-natural habitats and emphasises why it is so important that the mitigation hierarchy is adhered to so that impacts on such habitats occur only when there is no alternative.

5.15. The difficulty and uncertainty of successfully creating, restoring or enhancing a habitat is recognised in this multiplier.

[11] As the evidence base on the effectiveness of creation and restoration techniques grows and is reviewed the risk multipliers may be modified. A timetable for future updates/revision to the metric will be published.

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TABLE 5-1: Difficulty categories and multipliers

Difficulty categories	
Category	**Multiplier**
Very High	0.1
High	0.33
Medium	0.67
Low	1

Temporal risk

5.16. In delivering compensation there may be a mismatch in the timing of the impact and compensation, i.e. the difference in time between the negative impact on biodiversity and the compensation reaching the required quality. This results in lower levels of biodiversity for that period of time.

5.17. This issue can be managed by the creation of compensation habitat ahead of the impact taking place: e.g. by starting the offset work well ahead of the development for projects with a long lead in or through the creation of a bank of habitat units.

5.18. However, this is not always possible and even where the management to create compensation habitat starts in advance, the time taken for habitats to mature means that there will almost inevitably be a time lag. Where a time lag does occur, a risk multiplier is applied. This is referred to as the '**Time to target condition'** multiplier.

5.19. The time period to use in applying the Time to Target Condition multiplier to a metric calculation is the length of time (in years) between the intervention and the point in time the habitat reaches the pre-agreed target quality (i.e. distinctiveness, condition, area). This time will vary between habitat types, between change scenarios (e.g. creation typically takes longer than enhancement) and due to way the habitat is managed. Time to target condition values – based on based on good practice and typical conditions are provided for all habitats used in biodiversity metric 2.0. These values are set out in detail in the Technical Supplement.

5.20. These time to target condition values then need to be discounted. Discounting over time is an economic technique used to compare costs and benefits that occur in different time periods based around the principle that, generally, people prefer to receive goods and services now rather than later. Where time discounting is used in compensation schemes a standard discount rate is typically used. The biodiversity metric 2.0 uses 3.5%, which is the value recommended in the Treasury Green Book[12](Table 5-2 shows the multipliers for a number of time periods using a discount rate of 3.5 %. It is important to use precise figures (at least to 3 decimal places).

5.21. To be practical, the metric:

- **assumes that there is a quality 'jump' from the baseline condition to the target condition** once the relevant number of years has elapsed. Metric

[12] more details on discounting can be found in the Treasury Green Book Guidance, HM Treasury, 2011).

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calculations do not take into account incremental increases in quality of the habitat and do not need to be re-done annually, and

- **sets a limit on the discount rate used for temporal risk**. The metric sets a multiplier limit of x0.320 to take account of temporal risk. This equates to approximately 32 years, which is about the maximum time frame that most projects and plans can realistically plan ahead.

Monitoring is, however, recommended to confirm the actual number of biodiversity units delivered matches what was predicted.

TABLE 5-2: Time to target condition: multipliers for different time periods using a 3.5% discount rate.

Time to target condition			
Time (years)	**Multiplier**	**Time (years)**	**Multiplier**
0	1.000	17	0.546
1	0.965	18	0.527
2	0.931	19	0.508
3	0.899	20	0.490
4	0.867	21	0.473
5	0.837	22	0.457
6	0.808	23	0.441
7	0.779	24	0.425
8	0.752	25	0.410
9	0.726	26	0.396
10	0.700	27	0.382
11	0.676	28	0.369
12	0.652	29	0.356
13	0.629	30	0.343
14	0.607	31	0.331
15	0.586	>32	0.320
16	0.566		

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Off-site risk multiplier

5.22. There are both ecological and social drivers for compensation habitat to be provided local to where losses occur: e.g. the cultural ecosystem services provided by an area of land to a community. When off site compensation is within the local planning authority area (LPA) or the same National Character Area (NCA)[13] it is considered

[13] Further information on NCAs can be found at: http://publications.naturalengland.org.uk/category/587130

that those drivers have been addressed. However there is a risk of compensation being delivered at distance from the impact site. Where this is the case the off-site risk multiplier is applied to those compensation parcels outside of the relevant LPA or NCA. This risk is applied to area habitat, hedgerow and river elements of biodiversity metric 2.0. Note – for rivers and streams different off-site risk multipliers called riparian loss. See Chapter 8 for more details.

TABLE 5-3: Off-site risk categories

Off-site risk categories			
Category	**Score**	**Point applied to calculation**	
		Pre-impact	**Post-impact**
Compensation inside LPA or NCA of impact site	1.0	No	**Yes**
Compensation outside LPA or NCA of impact site but in neighbouring LPA or NCA	0.75	No	**Yes**
Compensation outside LPA or NCA of impact site and beyond neighbouring LPA or NCA	0.5	No	**Yes**

Biodiversity change scenarios

5.23. Different biodiversity change scenarios carry different levels of risk and the multipliers are applied differently to reflect this. Three distinct biodiversity habitat change scenarios (illustrated in Figure 3.1) are recognised in the biodiversity metric 2.0:

- **Habitat creation or recreation.** Where one habitat type is replaced by another or the habitat is destroyed (e.g. by development works) and the same habitat is recreated.
- **Habitat restoration or enhancement** of an existing habitat to improve its distinctiveness and / or condition. An example of restoration would be the transformation of a derelict chalk grassland dominated by scrub and coarse grasses to a continuous area of chalk grassland with isolated woody species and an abundance of fine-leaved grasses.
- **Accelerated habitat succession**. This recognises that certain interventions are comparable with ecological succession processes which result in a more distinctive habitat type (for example, grassland changing into scrub and ultimately woodland).The biodiversity value of the original habitat is not abruptly lost, but gradually changes as the new habitat type emerges. Accelerated succession interventions are subject to 'trading down' principles. Accelerated succession is a purposeful sustained intervention and it is envisaged that there are a limited number of situations where this would apply. For example, the planting of an existing grassland with thorny shrubs to facilitate natural tree regeneration to establish a woodland without removing the grassland.

5.24. Under the above scenarios different portions of the biodiversity value of a habitat may have different risk multipliers applied to it. So, as illustrated in Figure 5-1, in the

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case of a straightforward habitat creation, you lose all the original habitat, so the risks apply to the whole value of the habitat being created. Whilst in the case of restoration or enhancement the habitat starts with and retains a certain biodiversity value that interventions increase. The risk multipliers are applied to this uplift (improvement) of the habitat. In accelerated succession the situation is more complex. Recognising that the original habitat retains a biodiversity value while the new habitat emerges the metric applies risk only to the uplift in value resulting from succession.

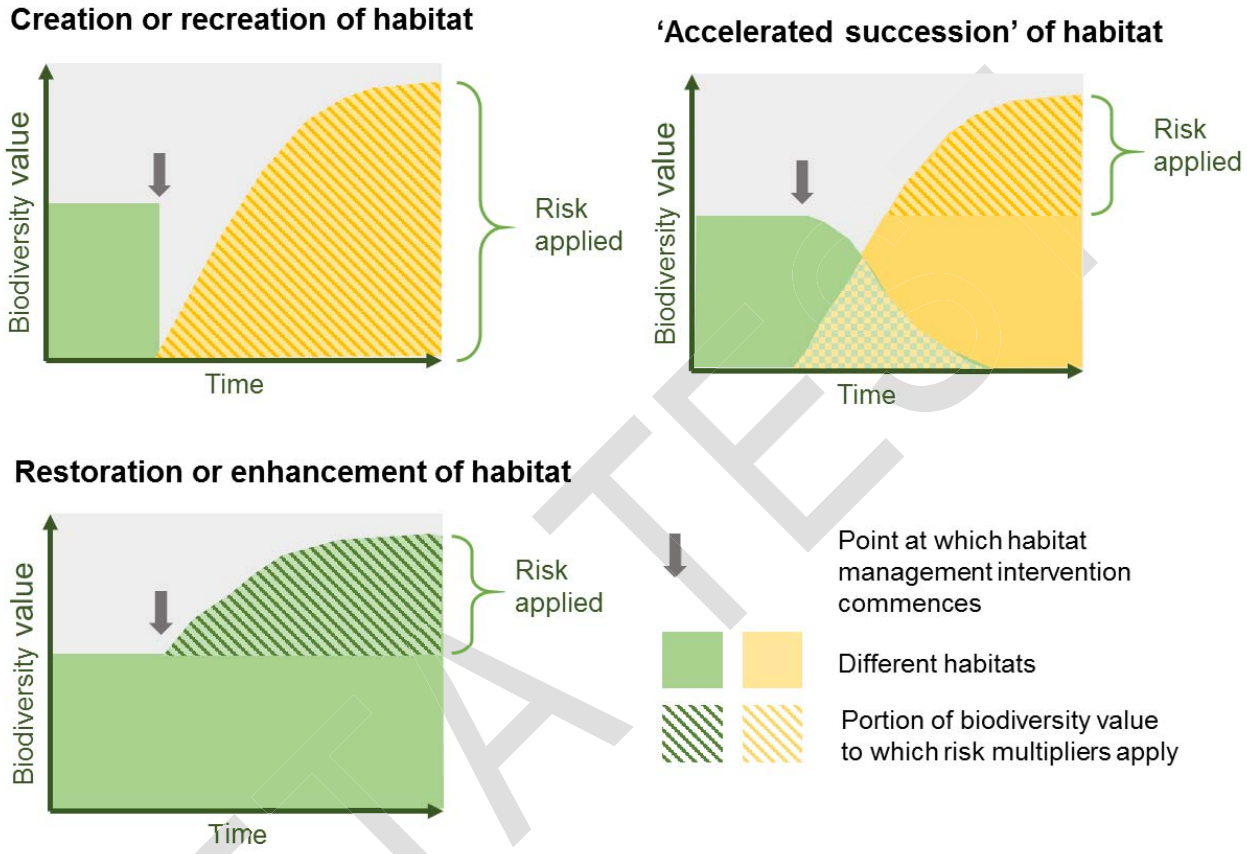


FIGURE 5.1: The biodiversity habitat change scenarios recognised in the metric

5.25. This leads to three different equations being used to generate biodiversity unit values pre and post intervention.

5.26. It is important to select the appropriate change scenario for each management intervention. This choice is an ecological judgement and is determined by the ecological consequences of the change, not where the habitat is located.

5.27. Compensation habitats can be created, restored or enhanced, or subject to accelerated succession on-site as well as off-site. Measures taken to generate biodiversity units by improving existing habitats must provide a significant and demonstrable uplift in distinctiveness or condition.

5.28. Good management practice does not, by itself, constitute restoration or enhancement, or accelerated succession.

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The spatial component

5.29. In biodiversity metric 2.0 there are two core spatial components. First, the **strategic significance** of a place for biodiversity, its geography. Second, ecological **connectivity**, the relationship of a habitat in a defined place to its immediate surroundings in respect of biological and ecosystem flows. While these concepts are not completely independent of each other they do represent different qualities of a habitat.

Strategic significance

5.30. The idea of strategic significance works at a landscape scale. It gives additional unit value to habitats that are located in preferred locations for biodiversity and other environmental objectives. Ideally these aspirations will have been summarised in a local strategic planning document which articulates where biodiversity is of high priority and the places where it is less so. Strategic significance utilises published local plans and objectives to identify local priorities for targeting biodiversity and nature improvement, such Nature Recovery Areas, local biodiversity plans, National Character Area[14] objectives and green infrastructure strategies. Table 5-5 shows the multiplier scores for both impact and compensation sites based on its place in a strategic plan.

5.31. In the absence of a locally or nationally relevant strategic documentation indicating areas of significance for biodiversity, the value of **1** should be used in pre and post development calculations. Use of a score of 1 does not penalise a proposal.

TABLE 5-5: Strategic significance categories and scores

Strategic Significance categories			
Category	**Score**	**Point applied to calculation**	
		Pre-impact	**Post-impact**
High strategic significance High potential & within area formally identified in local policy	1.15	**Yes**	**Yes**
Medium strategic significance Good potential but not in area defined in local policy	1.1	**Yes**	**Yes**
Low Strategic Significance Low potential and not in area defined in local policy	1	**Yes**	**Yes**

Connectivity

5.32. The focus of connectivity in biodiversity metric 2.0 is the relationship of a particular habitat patch to other surrounding **similar** or **related** semi-natural habitats. These help facilitate flows of species and ecosystem services increases habitat resilience.

[14] For more details of National Character Areas see: https://www.gov.uk/government/publications/national-character-area-profiles-data-for-local-decision-making/national-character-area-profiles

By **similar** habitats we mean, for example, multiple patches of calcareous grassland. By **related** habitats we mean habitats often found in association as part of a dynamic complex, for example lowland heath and scrub. The same approach is applied to impact and compensation sites.

5.33. In the beta version of biodiversity metric 2.0 all High and Very High distinctiveness habitats should be assigned a Medium connectivity multiplier, other habitats a Low connectivity multiplier (see Table 5-5).

5.34. A connectivity tool will be published in an updated version of biodiversity metric 2.0. which will use an approach based upon the habitat fragmentation or 'structural connectivity' model with the National Biodiversity Climate Change Vulnerability Model (NBCCVM)[15] to assess connectivity and will generate connectivity categories from highly connected to low connectivity. It encompasses the ideas of:

- Larger habitat patches being less susceptible to extreme events;
- Accommodation of a wider range of soil types, topography and microclimate affords greater niche variation;
- Potential for species dispersal and local re-colonisation to be facilitated; and
- Patch size and permeability of surrounding landscape being important for persistence of biodiversity.

TABLE 5-5: Beta version Connectivity multipliers assigned by habitat distinctiveness.

Connectivity Multipliers		
Habitat distinctiveness	**Connectivity**	**Multiplier**
Very high distinctiveness	Medium	1.1
High distinctiveness	Medium	1.1
Medium and low distinctiveness	Low	1

Moderating the influence of spatial components

5.35. So that strategic significance and connectivity elements do not have a disproportionate effect on the calculation outputs the specific scores are restricted in range in the beta version of biodiversity metric 2.0 to: strategic significance 1 – 1.15, and connectivity 1 – 1.1.

6: Area Habitat biodiversity unit calculations

6.1. Areas habitats are perhaps the most familiar ecological currency in the UK, they are the woodlands, grasslands, wetlands and other types that are widely recognised by ecologists and the public alike. The habitats we recognise comprise a community of different species populations living in a place. There is usually a sub-group of those

[15] For more information about the NBCCVM see: TAYLOR, S., KNIGHT, M. & HARFOOT, A. 2014. National biodiversity climate change vulnerability model. Natural England Research Report NERR054. Natural England. ISBN 978-1-78354-084-6.

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EFFECTS OF HELICOPTER NOISE ON MEXICAN SPOTTED OWLS

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Abstract: Military helicopter training over the Lincoln National Forest (LNF) in southcentral New Mexico has been severely limited to protect nesting Mexican spotted owls (*Strix occidentalis lucida*). To evaluate nesting and nonnesting spotted owl responses to helicopter noise, we measured flush frequency, flush distance, alert behavior, response duration, prey delivery rates, female trips from the nest, and nest attentiveness during manipulated and nonmanipulated periods, 1995–96. Chain saws were included in our manipulations to increase experimental options and to facilitate comparative results. We analyzed stimulus events by measuring noise levels as unweighted one-third-octave band levels, applying frequency weighting to the resultant spectra, and calculating the sound exposure level for total sound energy (SEL) and the 0.5-sec equivalent maximum energy level ($LEQ_{max\ 0.5\text{-}sec}$) for helicopters, and the 10-sec equivalent average energy level ($LEQ_{avg.\ 10\text{-}sec}$) for chain saws. An owl-weighting (dBO) curve was estimated to emphasize the middle frequency range where strigiform owls have the highest hearing sensitivity. Manipulated and nonmanipulated nest sites did not differ in reproductive success ($P = 0.59$) or the number of young fledged ($P = 0.12$). As stimulus distance decreased, spotted owl flush frequency increased, regardless of stimulus type or season. We recorded no spotted owl flushes when noise stimuli were >105 m away. Spotted owls returned to predisturbance behavior within 10–15 min after a stimulus event. All adult flushes during the nesting season occurred after juveniles had left the nest. Spotted owl flush rates in response to helicopters did not differ between nonnesting (13.3%) and nesting seasons (13.6%; $P = 0.34$). Spotted owls did not flush when the SEL noise level for helicopters was ≤102 dBO (92 dBA) and the LEQ level for chain saws was ≤59 dBO (46 dBA). Chain saws were more disturbing to spotted owls than helicopter flights at comparable distances. Our data indicate a 105-m buffer zone for helicopter overflights on the LNF would minimize spotted owl flush response and any potential effects on nesting activity.

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Key words: chain saws, disturbance, flush response, helicopters, Mexican spotted owls, noise, response thresholds, sound measurements, *Strix occidentalis lucida.*

To maintain tactical proficiency for low-level search-and-rescue missions, military air crews require frequent training. Recently, low-level training flights have come under scrutiny for their potential effects on wildlife, which has led to reductions in military access to potential training areas (Holland 1991). Holloman Air Force Base (HAFB), located in southcentral New Mexico, lacks sufficient area and habitat diversity to conduct effective helicopter training operations, but the Sacramento Ranger District of the LNF in the Sacramento Mountains contains the habitat diversity necessary to conduct such training operations. However, to avoid potential effects on nesting Mexican spotted owls (hereafter, spotted owl), military helicopters have been precluded from flying over the LNF during the February–August nesting season. To gain year-long access to the forest for training of the 48th Rescue Squadron (48 RQS), HAFB had to determine if and to what extent their activities might affect nesting spotted owls.

Much of the information about noise effects on raptors is anecdotal and fails to quantitatively measure either the stimulus or a behavioral response related to the animal's fitness. Predictive models for the relation between disturbance dosage and quantifiable effects are even more scarce (Awbrey and Bowles 1990, Grubb and Bowerman 1997). Although many types of human disturbance can affect birds of prey (Fyfe and Olendorff 1976), very little research has addressed the effects of human activity on owls (Wesemann and Rowe 1987; J. C. Bednarz and

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T. J. Hayden. 1988. The Los Medanos cooperative raptor research and management program, unpublished. Final Report 1985–87 for Department of Energy and Bureau of Land Management. University of New Mexico, Albuquerque, New Mexico, USA.). Presently, there is no published research available on the possible effect of noise on spotted owls.

The objectives of this study were (1) record, characterize, and quantify helicopter overflights at spotted owl roost sites during a post- or nonnesting season (1995) and at active nest sites during the nesting season (1996); (2) develop a dose-response threshold relation for quantifying spotted owl behavioral responses to variation in noise levels and stimulus distances; (3) determine if helicopter overflights affect spotted owl reproductive success (successful nests) or productivity (young fledged); and (4) develop disturbance-specific management guidelines to minimize potential effects from helicopter overflights on the LNF.

STUDY AREA

This study was located within the Sacramento Ranger District of the LNF in southcentral New Mexico, Otero County. This area was chosen for its large population of spotted owls and its importance as a potential training site for the 48 RQS. Vegetation is primarily Rocky Mountain conifer forest (Brown and Lowe 1980) dominated by Douglas-fir (*Pseudotsuga menzii*) with some southwestern white pine (*Pinus strobiformes*) and ponderosa pine (*P. ponderosa*; Alexander et al. 1984). Elevation in the mountainous terrain ranges between 1,372 and 2,957 m.

METHODS

Surveys and Territory Selection

Territories were surveyed between 15 June and 6 July 1995 and between 15 March and 15 April 1996. We selected territories for our study based on (1) presence of mated pairs of spotted owls, and (2) no captures or manipulations prior to our research. During the 1996 nesting season, a third criterion required pairs to be reproductively active. Seven female and 6 male spotted owls were tested in the nonnesting season. During the nesting season, we concentrated on testing females because of their nest fidelity. Sample sizes in both the nonnesting and nesting seasons were limited by the number of pairs that fit the foregoing criterion.

Vocal imitations of spotted owl calls (Forsman 1983) were used during nocturnal and diurnal point surveys to locate both nonnesting and nesting spotted owls. During nocturnal surveys, we determined initial spotted owl locations between dusk and 2200 (all times reported as Mountain Standard Time). Spotted owl positions were triangulated by plotting compass bearings on topographic maps so that each area could be visited for diurnal surveys (predawn to 0800), when we followed spotted owls to a daytime roost. During the 1995 nonnesting season, we then radioed the location to the 48 RQS for an overflight later that day or conducted a chain saw manipulation. Upon finding a spotted owl during the 1996 nesting season, we attempted to determine its reproductive status by feeding it live mice. Nesting spotted owls take prey back to the nest, while nonreproductive spotted owls either cache or eat the prey (Forsman 1983). Once a territory was determined to be reproductively active, the nest location was recorded for future testing. To minimize interactions with spotted owls, we used the least number of mice necessary to determine reproductive activity.

Spotted Owl Behavior and Response Measures

We documented spotted owl behavior during manipulated and nonmanipulated periods by direct observation (camouflaged blinds 25–30 m from nest or roost) and through video surveillance. To evaluate spotted owl response behavior to helicopter and chain saw manipulations and contrast it with pre- and postmanipulated behavior, we measured the following: (1) flush frequency = proportion of manipulations that elicited a flight response; (2) flush distance = distance (m) flown by a spotted owl in response to a sound stimulus; (3) alert behavior = number of head movements averaged per 5-min block, before, during and after manipulations; (4) time to alert = minutes between start of a manipulation and when a spotted owl initially responded with a head movement in the direction of the manipulation; (5) response duration = minutes following a noise stimulus until a disturbed spotted owl returned to prestimulus behavior; (6) prey deliveries = number of prey deliveries recorded at each nest site (calculated per hour for diurnal, nocturnal, and 24-hr periods); (7) trips = number of times the attending female left the nest (calculated per hour for diurnal, nocturnal, and 24-hr periods); and (8)

nest attentiveness = proportion of time the adult female spotted owl spent on the nest through the nesting season (calculated for diurnal, nocturnal, and 24-hr periods, as well as for nesting phases; see Delaney et al. 1999).

Video Surveillance

Because our use of video was a new and unique application of the various hardware components, we had to design, construct, test, and modify our video surveillance system before applying it in the field (Delaney et al. 1998). We used Marshall black-and-white, miniature video cameras (Marshall Electronics, Culver City, California, USA) with night vision to monitor spotted owl behavior. (Use of trade names does not imply endorsement by the U.S. Forest Service Rocky Mountain Research Station, U.S. Air Force, U.S. Army Construction Engineering Research Laboratories, or Northern Arizona University to the exclusion of other potentially suitable products.) In addition to the 6 infrared light-emitting diodes (LEDs) on the camera board, 9-LED supplemental light sources were constructed to approximately double night-vision capabilities. Panasonic industrial-grade video recorders provided up to 24-hr coverage/VHS tape (Panasonic Corporation of America, Secaucus, New Jersey, USA). Between 9 April and 27 May 1996, cameras were placed at 20 nest sites in adjacent trees, averaging 6.9 m from nests (range = 3.0–10.3 m). A 15-m, power-line-and-coaxial-cable down line and a 60-m trunk line were used to minimize potential disturbance to spotted owls by offsetting the recorder and batteries to an out-of-sight tarpaulin blind. Between 25 April and 3 July 1996, video surveillance systems at 19 successful nests yielded >2,655 hr of spotted owl behavior coverage. All cameras and related equipment were removed after the 1996 nesting season.

Sound Measurements

Instrumentation and Recording.—Sony TCD-D7, digital audio tape (DAT) recorders continuously recorded all noise events, along with exact time and date (Sony Corporation of America, New York, New York, USA). We attached a Bruel & Kjaer (B&K) Type 4149, 1.3-cm condenser microphone (Bruel & Kjaer, Naerum, Denmark) with a 7.5-cm wind screen to a B&K Model 2639 preamplifier, mounted the microphone on a 1-m stick, and placed the unit directly under a spotted owl location (roost or nest) about 1 m from the tree trunk. Using 3 10-m connecting cables attached to the preamplifier, we located the B&K Model 2804 power supply and DAT recorder at our observation point in a camouflaged blind 25–30 m from the spotted owls. A 1.0-kHz, 94-dB calibration signal from a B&K Type 4250 sound level calibrating system was recorded before and after each recorded manipulation. This signal provided an absolute, standardized reference point for sound levels and spectra when data were later reduced via a B&K Type 2144 frequency analyzer. All noise data were analyzed at the U.S. Army Construction Engineering Research Laboratories, Champaign, Illinois, USA.

Sound Metrics.—We used 3 sound metrics in this study: (1) SEL = the sound exposure level, which represents total sound energy for helicopters; (2) $LEQ_{avg.\ 0.5\text{-}sec}$ = the 0.5-sec equivalent average peak energy level for helicopters; and (3)$LEQ_{avg.\ 10\text{-}sec}$ = the 10-sec equivalent average energy level for chain saws. Noise is defined as sound that is undesired or that constitutes an unwarranted disturbance; it can alter animal behavior or normal functioning. We analyzed noise events as unweighted one-third-octave band levels, applied frequency weighting to the resultant spectra, and calculated the above metrics. Chain saw noise was relatively steady and most appropriately described by the average sound level, LEQ, over a specified time interval (10-sec). Helicopter noise was more varied and could not be described as well by the average noise level. Aircraft noise events are typically described in terms of SEL, which correlates well with human annoyance to aircraft noise. However, SEL levels cannot be meaningfully compared to LEQ levels; therefore, we also represented helicopters with LEQ for comparison with chain saws.

Frequency weighting is an algorithm of frequency-dependent attenuation that simulates the hearing sensitivity of the study subjects. Frequency weighting discriminates against sound which, while easily measured, is not heard by the subjects. Flat-weighting (or absence of any weighting function) does not emphasize any portion of the frequency spectrum and therefore represents the true sound level and frequency for a noise stimulus event (Fig. 1). The commonly used A-frequency weighting attenuates noise energy according to human hearing range and sensitivity and generally will not be appropriate for animal species. However,

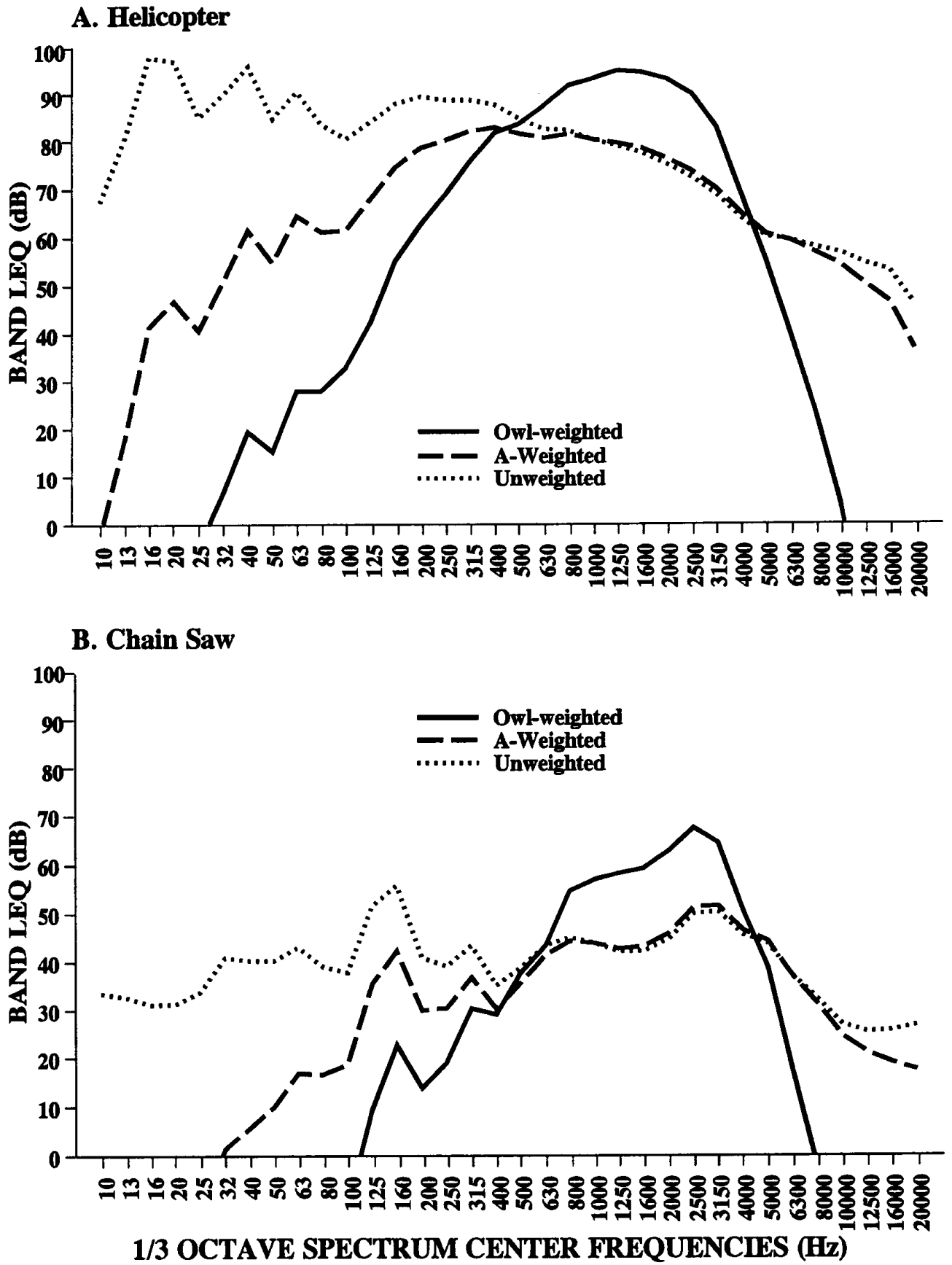


Fig. 1. (A) A comparison of owl-, A-, and unweighted equivalent maximum (helicopter) and average (chain saw) noise energy levels (LEQs) for a 60-m helicopter manipulation on 5 June 1996, and (B) a 60-m chain saw manipulation on 11 June 1996 at the same Mexican spotted owl territory in the Sacramento Mountains, New Mexico.

Table 1. Distribution of sample size by season, manipulation type, and Mexican spotted owl nest sites for noise-effect testing in the Sacramento Mountains, New Mexico, 1995–96. Site totals are not necessarily additive because some sites were manipulated in both years, and not all sites received both helicopter and chain saw manipulations.

Season	Helicopter		Chain saws		Season totals	
	Manipulations	Sites	Manipulations	Sites	Manipulations	Sites
Nonnesting (1995)	24	8	25	13	49	13
Nesting (1996)	57	22	55	21	112	22
Totals	81	26	80	27	161	28

it is useful to present A-weighted noise levels (dBA), as well as more appropriately weighted levels, because this weighting algorithm occurs on sound-level meters and is ubiquitously used.

Because both flat- and A-weighting do not accurately reflect the way a spotted owl hears noise, we developed an estimated owl-weighting (dBO) curve. An audiogram describes hearing range and sensitivity and provides information on which a frequency weighting algorithm can be based for a specific species. Available information indicates hearing is quite similar among members of a taxonomic order. Within the order Strigiformes, we found audiograms for 2 species (great horned owl [*Bubo virginianus*], barn owl [*Tyto alba*]) within the same Suborder (Strigi) as spotted owls. These audiograms were used to approximate frequency-weighted noise levels for spotted owls. This owl-weighting emphasized the middle frequency range where test spotted owls had the highest hearing sensitivity (Trainer 1946, Konishi 1973).

Field Manipulations

We conducted a pilot test in January 1995 on HAFB and at an inactive nest site on the LNF to determine experimental flight profiles and microphone placement. Helicopter manipulations occurred between 1 and 22 August 1995 and between 30 April and 25 July 1996. Chain saw manipulations were conducted between 9 July and 23 September 1995 and between 11 June and 26 July 1996. We manipulated 28 spotted owl territories (13 in 1995 and 22 in 1996, with 7 sites manipulated in both years). Twenty-five of these sites received both helicopter and chain saw manipulations, while the remaining 3 sites received only 1 chain saw or helicopter manipulation (Table 1). We tried to minimize the overall number of manipulations and to maximize the time between manipulations, while still striving to conduct as complete an array of manipulations as possible during the incubation, nestling, and fledgling phases of the nesting season. However, because of administrative and logistical delays, as well as to remain conservative in our approach, only 3 chain saw tests and 8 helicopter flights were conducted during incubation. The average interval between consecutive manipulations, regardless of type or season, was 12.8 days (range = 4–79).

During the 1995 field season, spotted owls were manipulated after the normal nesting cycle so that any behavioral responses could not have an adverse effect on nesting success or productivity. Only after spotted owls showed minimal responses during nonnesting did we focus manipulations on nesting spotted owls. Helicopter manipulations were comparable between seasons but, as explained below, a threshold validation approach was taken with chain saws during the 1996 nesting season to limit potential experimental effects. Therefore, we could only compare helicopter and chain saw results for the nonnesting season. This research was conducted under a U.S. Forest Service subpermit to the U.S. Fish and Wildlife Service Region 2 Endangered Species and Special Purpose-Master-Migratory Bird Permits.

Helicopters.—Helicopter tests were conducted with the actual aircraft used by the 48 RQS (Sikorsky, HH-60G, Pave Hawk, twin-jet helicopters). The blade design of the HH-60G greatly reduces blade-slap ('whopping' sound) of an approaching helicopter. Spotted owl territories were randomly presented with 1 of 3 controlled helicopter flight profiles on any 1 day: (1) 15-m vertical, (2) 30-m vertical–30-m lateral, and (3) 60-m vertical. All flights were above tree canopy. The 15-m flight represented a minimum altitude that 48 RQS pilots would rarely fly during training missions. The 60-m profile represented the maximum diurnal altitude and a minimum nocturnal flight altitude. The 30-m vertical–30-m lateral profile approximated a more typical daytime overflight. At each site, we tried to conduct all 3 diurnal profiles during both the nonnesting and nesting

seasons, plus at least 1 nocturnal profile during the nesting season.

Terrain, tree heights, and variation among pilots caused deviations from the intended profiles. To account for these variations and to facilitate a comparison with chain saw results, we calculated straight-line distances from spotted owl to helicopter for all aircraft manipulations. We used field observations of aircraft from 2 to 3 positions of varying elevation and lateral offset, measured tree and spotted owl heights, topographic features, pilot information (Global Positioning System [GPS] flight path data, aircraft altitude, crew observations), and triangulation for these calculations, which we estimated to be ± 10% accurate. In addition to calculating closest distance, we also calculated spotted owl-to-aircraft distance during the approach, using flight speed data to determine the distance at which spotted owls first responded to approaching helicopters. We conducted only 1 pass/flight over any roost or nest site per day, with the entire manipulation lasting <10 min in total audible duration and <30 sec in the immediate vicinity of the spotted owls. Helicopter speed was 150–170 km/hr (80–90 knots).

To define the specific flight line, we positioned 2 1-m-diameter red, helium-filled weather balloons above the canopy approximately 50 m on either side of a spotted owl's position. To position nocturnal flights, we used flashlights (4 D-cell Mag Lites; Mag Instruments, Ontario, California, USA) pointed skyward in conjunction with the pilots' night-vision capabilities. Diurnal flights usually occurred between 1200 and 1300 and nocturnal flights between 2000 and 2200. The 48 RQS developed a pilot's in-flight guide which detailed all spotted owl territories and access routes so pilots could circumvent nontarget and control sites en route to manipulated sites.

Chain Saws.—Stihl Model 025, 44-cc chain saws were used for noise testing (Stihl, Virginia Beach, Virginia, USA). To satisfy LNF fire and safety restrictions, bars and chains were removed and there was no actual cutting during manipulation testing. However, the noise levels and frequency spectra were similar for chain saws tested with and without bars and chains. We used forest vegetation to hide the operator and eliminate visibility as an influencing factor as we ran chain saws continuously for 5 min, alternately revving for 10 sec and idling for 10 sec. In 1995, nonnesting spotted owls were exposed to chain saw noise from 1 of 5 randomly selected initial distances (30, 45, 60, 75, and 90 m). If spotted owls flushed during the initial presentation, the test was ended for that day and the next scheduled manipulation was initiated 15–30 m farther away to establish a distance-response threshold. If the initial manipulation did not cause a flush, the next manipulation about 5–7 days later was presented 15 m closer. Because 60 m was the greatest distance at which chain saws elicited a flush response in 1995, only distances ≥60 m were examined in 1996 (60, 105, 250, and 400 m). This approach minimized potential effects on nesting spotted owls.

Habituation

We used experimental testing and treating sites as their own controls to evaluate possible habituation to repeated noise stimuli. At the end of the 1996 nesting season, we exposed 4 previously unmanipulated sites and 4 manipulated sites to 1 helicopter and 1 chain saw manipulation each, and we compared spotted owl flush response. Helicopter flights followed the most aggressive 15-m profile, while chain saw tests were run at 60 m to remain consistent with our conservative approach to nesting season manipulations. In addition, manipulated sites were used as their own self-controls throughout the study. We measured temporal changes in spotted owl response toward disturbance based on seasonal (response duration and time-to-alert) as well as proximate scales (alert responses pre-, during, and postmanipulation).

Data Analyses

Numbers of manipulated sites and sample sizes for individual analyses varied due to different inclusion criteria, missing data, 7 sites being manipulated in both years, and not all sites receiving both helicopter and chain saw manipulations. We used SPSS 7.5 for Windows (SPSS 1997) to perform descriptive statistics, independent-samples t-test for comparing mean values of young fledged and variation in sound levels, Tukey's HSD test in the 1-way analysis of variance for comparing distances for alert, react, and flush responses, and linear regression for exploring the relation between noise levels and distance by type and between net differences in prey deliveries before and after manipulations. We used net differences in prey deliveries because of repeated-measures limitations. We cal-

culated a potential threshold distance for zero-difference with a 95% calibration interval (Graybill 1976). We used a 1-tailed Fisher's exact test to assess 2 × 2 contingency tables for flush response variability with manipulation type, stimulus distance, nesting season and phase, and for reproductive success between experimental and control sites (Zar 1984). To evaluate mean alert response (i.e., head movements) for 5-min intervals pre-, during, and postmanipulation, we used a nonparametric, Multi-Response Permutation Test for matched pairs (PTMP; Mielke and Berry 1982; Slauson et al. 1991. User manual for BLOSSOM statistical software, unpublished. National Ecological Research Center, U.S. Fish and Wildlife Service, Fort Collins, Colorado, USA). We used power analyses (Steidl et al. 1997) on reproductive success and productivity for comparisons between experimental and control sites. All means are reported ± SD. We considered alpha levels of $P < 0.05$ significant.

RESULTS

Manipulation Summary

We presented 161 helicopter and chain saw manipulations during the 1995 nonnesting and the 1996 nesting seasons (Table 1). The fledgling phase received 30 helicopter and 43 chain saw manipulations. The nestling phase received 19 helicopter and 9 chain saw manipulations, and the incubation phase received 8 helicopter and 3 chain saw manipulations. We were not able to compare spotted owl response levels between diurnal and nocturnal periods because, due to scheduling and logistical difficulties, there were only 5 nocturnal helicopter flights over spotted owl nests during the nesting season. Because we analyzed only first exposures (when >1 occurred) for each categorical manipulation distance at each site for flush response, our effective sample size by distance was reduced to 126: 58 helicopters (4 at ≤30 m, 13 at 30–45 m, 15 at 46–60 m, 20 at 61–105 m, 6 at >105 m) and 68 chain saws (6 at 30 m, 3 at 45 m, 23 at 60 m, 3 at 75 m, 2 at 90 m, 16 at 105 m, 13 at 250 m, 2 at 400 m). The 8 helicopter and 8 chain saw postexperiment habituation manipulations were in addition to this sample of 126.

Reproductive Success

Manipulated and nonmanipulated nest sites did not differ in reproductive success (Fisher's exact test: $P = 0.59$) or the number of young fledged ($t_{20} = 0.95$, $P = 0.12$). We conducted power analyses based on data from 17 experimental and 5 control sites. Power levels were 0.22 for detecting a 10% difference in reproductive success between nests and 0.11 for productivity. Power increased to 0.29 and 0.16 for detecting a 15% difference, and 0.36 and 0.21 for a 20% difference. Fifteen of 17 manipulated spotted owl nest sites produced 1.4 young/occupied nest (1.6 young/successful nest), while all 5 nonmanipulated sites were successful in producing 1.8 young/occupied nest and 1.8 young/successful nest. Neither of the failed pairs flushed nor exhibited any unusual response to manipulations during the nesting season. One pair lost their chick to predation 9 days after the last manipulation, while apparently infertile eggs at the other site never hatched, despite normal incubation.

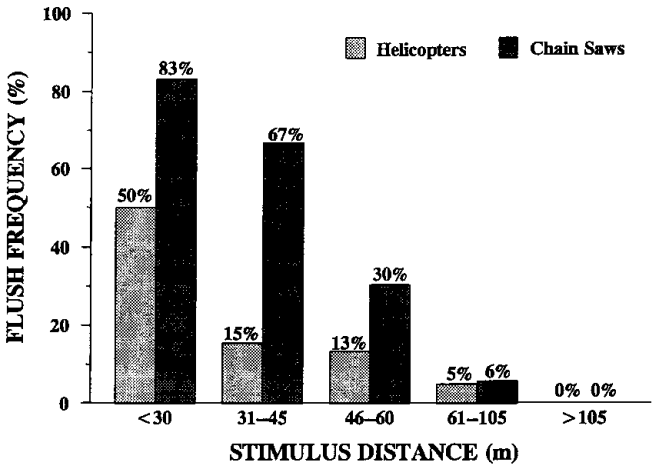


Fig 2. A comparison of Mexican spotted owl flush frequency by stimulus type and distance for helicopter and chain saw manipulations in the Sacramento Mountains, New Mexico, 1995–96.

Flush Response and Associated Thresholds

Distance Thresholds.—As stimulus distance decreased, spotted owl flush frequency increased (Fig. 2), regardless of stimulus type (Tables 2, 3) or season (nesting, nonnesting). We recorded no spotted owl flushes when the noise stimulus was >105 m distant. Only 2 flushes occurred at >60 m stimulus distance (1 helicopter at 89 m, 1 chain saw at 105 m). Chain saws consistently elicited higher response rates than helicopters at similar distances (Fig. 2). At ≤60 m stimulus distance during the nonnesting season, response to chain saws (72%) was greater than response to helicopters (20%; Fisher's

Table 2. Helicopter manipulations eliciting a Mexican spotted owl flush response during the 1995 nonnesting and 1996 nesting seasons in the Sacramento Mountains, New Mexico.

Season	Stimulus distance	Number of presentations	Number of flushes	Flush frequency
Nonnesting (1995)	≤30	1	1	100.0
	30–45	6	1	16.7
	46–60	8	1	12.5
	61–105	7	0	0.0
	≤105	22	3	13.6
	>105	0	0	0.0
1995 season totals		22	3	13.6
Nesting (1996)	≤30	3	1	33.3
	30–45	7	1	14.3
	46–60	7	1	14.3
	61–105	13	1	7.8
	≤105	30	4	13.3
	>105	6	0	0.0
1996 season totals		36	4	11.1
Helicopter totals		58	7	12.1

exact test: $P < 0.01$). In 11 instances at 6 territories (46% of the 24 helicopter flights during nonnesting), spotted owls did not flush in response to helicopter noise that averaged 21 dBO louder than chain saw manipulations that did cause a flush at the same territory (Fig. 3). All flushes recorded during the nesting season occurred after fledging; no flushes were elicited by manipulations during the incubation and nestling phases. Overall, helicopters elicited 0% spotted owl response when beyond 105 m, 14% within 105 m, 19% within 60 m, and 50% within 30 m.

Spotted owl flush rates did not differ between nesting (13.6%) and nonnesting (13.3%) seasons for helicopter manipulations at ≤105 m (Fisher's exact test: $P = 0.34$; Table 2). Flush rates were lower during the incubation and nestling phases (0%) than during the fledgling phase (28%; Fisher's exact test: $P = 0.04$). Adults also roosted farther from juveniles as the number of days postfledging increased (1–20 days: $\bar{x} = 9.7$ m, $n = 10$; 21–40 days: $\bar{x} = 18.2$ m, $n = 15$; 41–60 days: $\bar{x} = 29.3$, $n = 11$). After 20 days, adult flush distance was typically less than adult-to-juvenile distance (21–40 days: $\bar{x} = 10.8$ m, $n = 2$; 41–60 days: $\bar{x} = 13.7$, $n = 3$). An adult spotted owl flew closer to a juvenile during only 1 of these latter 5 manipulations; however, regardless of adult flush distances, new diurnal roosts averaged only 6.5 m farther from juveniles.

Table 3. Chain saw manipulations eliciting a Mexican spotted owl flush response during the 1995 nonnesting and 1996 nesting seasons in the Sacramento Mountains, New Mexico.

Season	Stimulus distance (m)	Number of presentations	Number of flushes	Flush frequency (%)
Nonnesting (1995)	30	6	5	83.3
	45	3	2	66.7
	60	9	6	66.7
	75	3	0	0.0
	90	2	0	0.0
	105			
	≤105	23	13	56.5
1995 season totals		23	13	56.5
Nesting[a] (1996)	60	14	1	7.1
	105	16	1	6.3
	250	13	0	0.0
	400	2	0	0.0
1996 season totals		45	2	4.4
Chain saw totals		68	15	22.1

[a] To minimize additional potential disturbance at nest sites, only chain saw distances ≥60 m were tested in 1996; therefore, season totals are not comparable.

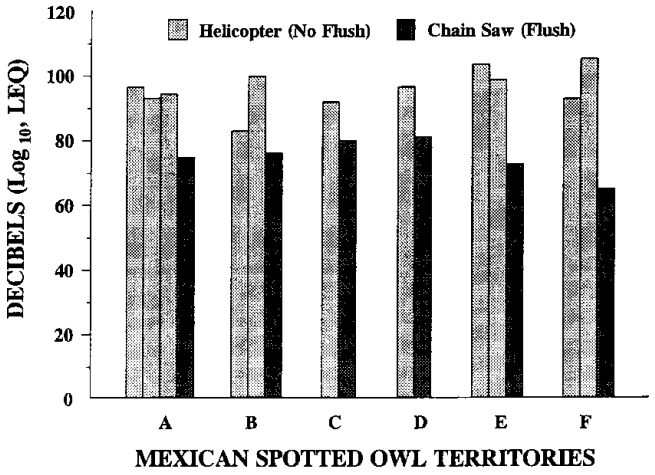


Fig. 3. A comparison of Mexican spotted owl flush response to chain saw and no-flush response to louder helicopters flown at equal or lesser distances at 6 territories during the 1995 nonnesting season in the Sacramento Mountains, New Mexico.

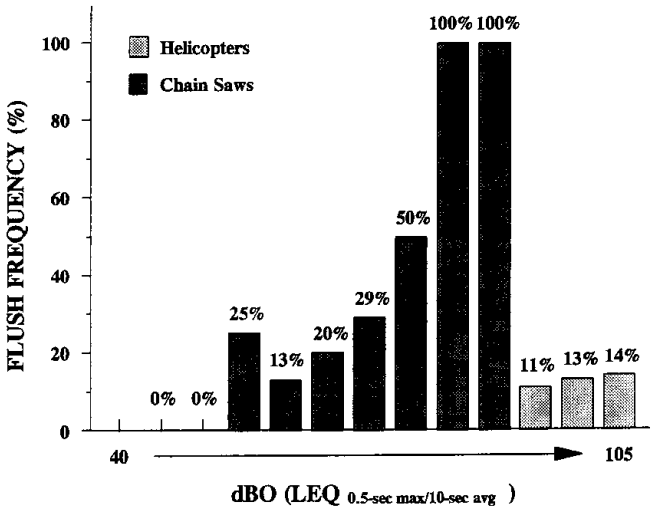


Fig. 4. A comparison of Mexican spotted owl flush frequency by stimulus type and noise level for helicopter and chain saw manipulations in the Sacramento Mountains, New Mexico, 1995–96.

Noise Thresholds.—During the nonnesting season, spotted owls did not flush when the SEL noise level for helicopters was ≤104 dBO (92 dBA) and the LEQ level for chain saws was ≤65 dBO (51 dBA). During the nesting season, spotted owls did not flush when the SEL sound level for helicopters was ≤102 dBO (92 dBA) and the LEQ level for chain saws was ≤59 dBO (46 dBA). These dB levels represent the noise level thresholds below which there were no spotted owl flush responses for the stimulus type and season indicated. Noise levels recorded near nest sites before and after disturbance trials were usually 25–35 dB (reaching upwards of 40 dB on windy days). Helicopters typically became audible at approximately 2,000 m.

Owl-, A-, and flat-weighting curves differed for 2 equidistant helicopter and chain saw trials at the same site (Figs. 1A,B). Within the midfrequency range, helicopters were louder than chain saws; yet, more of the total chain saw noise energy was in the midfrequency range where estimated spotted owl sensitivity was greatest. Helicopter energy level peaked at the lower end of the spectrum below the estimated spotted owl hearing sensitivity range. This difference partially explains the higher response rates for chain saws at lower noise levels than for helicopters (Fig. 4).

Alert Response

Spotted owls exhibited alert responses (i.e., head movements) when helicopters were an average of 403 ± 148 m away (n = 34; Fig. 5) but showed no response when helicopters were >660 m distant. Distances between helicopter and spotted owl for react responses (i.e., wing and body movements; $\bar{x}$ = 124 ± 59 m, n = 2) and flush responses (i.e., flight; $\bar{x}$ = 45 ± 28 m, n = 7) were shorter than for alert responses (Tukey's HSD: $P < 0.01$). The stimulus distances for react and flush responses did not differ (Tukey's HSD: $P = 0.75$), but sample sizes were very limited. However, the indicated trend was for severity of response type to increase as stimulus distance decreased.

We compared average head-movements/5 min for the 30–60 min prior to a manipulation (i.e., the mean of all 5-min, premanipulation means), the 5-min interval of the manipulation, and the 30–60 min following the manipulation. Spotted owls responded to noise stimuli with more alert movements ($\bar{x}$ = 7.4 ± 5.6, n = 91)

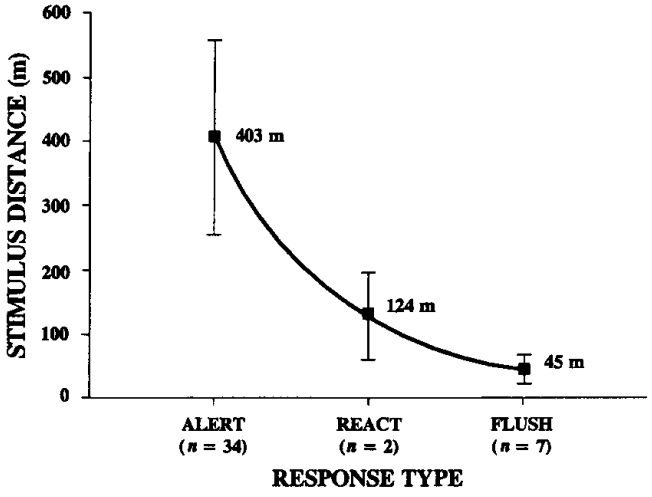


Fig. 5. Relation between stimulus distance and Mexican spotted owl response type during helicopter flights at 26 nest sites in the Sacramento Mountains, New Mexico, 1995–96. Error bars denote 2 standard deviations of the mean.

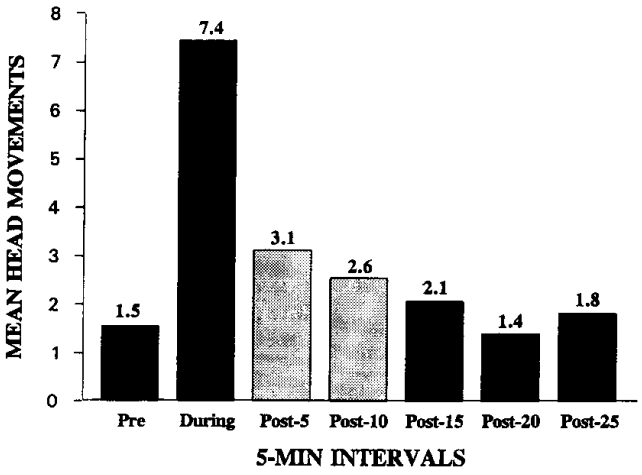


Fig. 6. Variation in mean frequency of Mexican spotted owl head movements (alert response) pre-, during, and postmanipulation for helicopter and chain saw noise events in the Sacramento Mountains, New Mexico, 1995–96. Bar shading indicates significant variation ($P \leq 0.08$).

than during premanipulation intervals ($\bar{x}$ = 1.6 ± 1.5, n = 84; PTMP: $P < 0.01$; Fig. 6). Spotted owls typically returned to premanipulation behavior in the third 5-min interval (between 10–15 min) after a stimulus event. Only the first (3.1 ± 3.7, n = 57; PTMP: P = 0.01) and second 5-min intervals (2.6 ± 3.0, n = 51; PTMP: P = 0.08) following a manipulation had greater frequencies of head movements than premanipulation levels.

Habituation

Experimental Testing.—Three of 4 previously unmanipulated spotted owls flushed in response to helicopter flights, while none of the 4 previously manipulated spotted owls flew (Table 4; Fisher's exact test: P = 0.07). During chain saw testing, 2 of the 4 unmanipulated spotted owls flushed, while no manipulated spotted owls flew (P = 0.21). Spotted owls may have habituated to the manipulations during successive exposures, and more so to helicopters than to chain saws. However, sample sizes were too small to establish significance for indicated trends.

Seasonal Change in Response Duration.—Response duration was consistently longer for chain saws than helicopters, was inversely related to stimulus distance, and generally decreased as the nesting season progressed. Mean response duration following helicopter flights dropped from 10.3 ± 9.4 min (n = 14) in July to 8.2 ± 5.5 min (n = 12) in August. Mean response durations following chain saw manipulations were 22.2 ± 22.3 min (n = 19) in July and 10.7 ± 8.6 min (n = 5) in August. Response durations following chain saw manipulations were 1.3–2.2 times longer than following helicopter flights.

Response duration averaged 16.6 ± 16.8 min (n = 47) when stimuli were ≤60 m away, and 7.0 ± 7.9 min (n = 25) when stimuli were >60 m away. Spotted owls required 11.6 ± 10.5 min (n = 24) to return to prior resting condition after helicopter flights of ≤60 m, compared to only 6.0 ± 8.8 min (n = 11) when flights were >60 m away. Response durations for chain saws were 21.1 ± 20.9 min (n = 23) at ≤60 m and 7.4 ± 7.6 min (n = 14) at >60 m, which was 1.2–1.8 times longer than helicopter durations.

Table 4. Habituation testing of helicopter and chain saw noise stimuli at 4 manipulated Mexican spotted owl nest sites and 4 previously unmanipulated sites in the Sacramento Mountains, New Mexico, 1996.

Parameter	Helicopter		Chain saw	
	Unmanipulated	Manipulated	Unmanipulated	Manipulated
Dates	8–25 Jul	8–22 Jul	19–23 Jul	16–26 Jul
Flushes	3 (75%)	0	2 (50%)	0
Distance (m, SLD)[a]				
Mean	36.5	42.8	60.0	60.0
Range	33–40	40–47	60.0	60.0
Owl-weighting (dB)[b]				
Mean	101.7	100.9	69.9	68.5
Range	[c]	99.8–102.3	64.9–72.9	65.2–74.5
A-weighting (dB)[b]				
Mean	95.5	90.4	56.4	56.0
Range	[c]	89.9–91.4	51.3–59.7	53.8–60.5

[a] SLD = straight-line distance between stimulus and spotted owl.
[b] dB = decibels.
[c] Only 1 noise level recording was available for analysis.

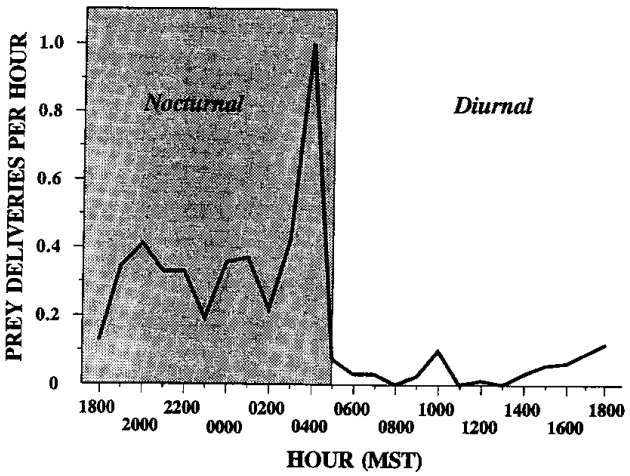


Fig. 7. Mean prey deliveries per hour (Mountain Standard Time [MST]) averaged across the entire nesting season for Mexican spotted owls in the Sacramento Mountains, New Mexico, 1996.

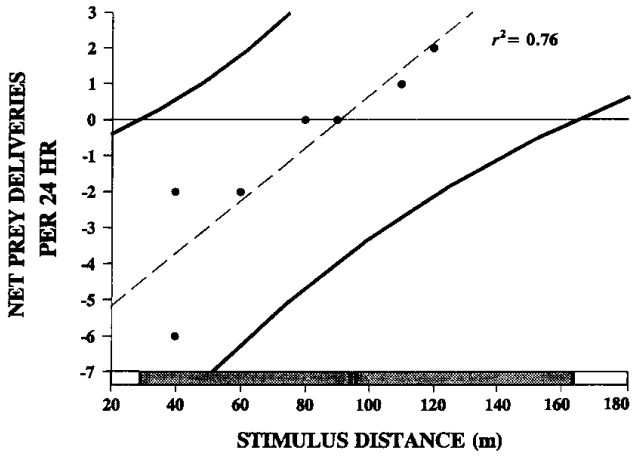


Fig. 8. The linear regression and 95% prediction interval between stimulus distance and net difference in prey deliveries (postmanipulation deliveries minus premanipulation deliveries) for the 24-hr periods after and before helicopter and chain saw manipulations at 4 Mexican spotted owl nest sites in the Sacramento Mountains, New Mexico, 1996. Shading along the distance axis indicates the 95% calibration interval (± 68 m) around the estimated potential threshold distance (96 m) for zero-difference in prey deliveries (Graybill 1976).

Seasonal Change in Time to Alert.—Time to alert increased as the nesting season progressed and as stimulus distance increased. Spotted owls exhibited alert responses after only 0.6 ± 0.7 min (n = 13) during helicopter flights in July, compared with 1.4 ± 1.4 min (n = 9) in August and 1.6 ± 0.6 min (n = 6) in September. Spotted owl response was quicker during chain saw manipulations: <0.2 ± 0.4 min (n = 22) in June, >0.2 ± 0.4 min (n = 13) in July, 0.3 ± 0.5 min (n = 6) in August, and 0.5 min (n = 1) in September. When helicopter flights were ≤60 m away, spotted owls responded in 1.0 ± 1.0 min (n = 21) compared to 1.2 ± 1.0 min (n = 10) for flights >60 m. Spotted owls responded to chain saws in 0.1 ± 0.2 min (n = 26) at distances ≤60 m, and in 0.3 ± 0.5 min (n = 16) when saws were >60 m away. Time to alert was consistently 3.0–10.0 times longer for helicopters than for chain saws.

Prey Delivery Rates and Related Behaviors

Over 81% of all prey deliveries within the nesting season (n = 387) occurred during nocturnal hours (n = 16 spotted owl territories). Mean prey deliveries per hour were highest (1.00) at 0400, when >18% of all prey were delivered (Fig. 7). Prey deliveries per hour averaged 0.03 during diurnal hours compared with 0.37 during nocturnal hours, which translates to 0.36 prey deliveries/12-hr diurnal period and 4.20 deliveries/12-hr nocturnal period.

There were only 7 instances of full 24-hr video records 1–2 days before and immediately after a manipulation. Net differences in prey deliveries for the 24-hr periods after and before noise manipulations (postmanipulation deliveries minus premanipulation deliveries; Fig. 8) were highly correlated with stimulus distance (r^2 = 0.76, n = 7 at 4 sites). The estimated potential threshold distance for a negative effect on prey deliveries was 96 m, which is consistent with the 105-m threshold for flush response described above. Because of limited sample size, the 95% calibration interval around this estimated threshold ranged between 28 and 164 m. Experimental helicopter and chain saw manipulations did not affect spotted owl nest attentiveness or the number of female trips from the nest; differences for the 24-hr periods pre- and postmanipulation were not correlated with stimulus distance.

DISCUSSION

Research Effects

Spotted owls tend to be less affected by nearby, nonthreatening human activity than most other raptor species. Sovern et al. (1994) found both nesting and nonnesting spotted owls became accustomed to observers sitting quietly 25–50 m away in only 10–15 min. Our use of blinds and their placement, along with the microphones, 1–4 hr in advance of manipulations provided additional visual and temporal buffering. Monitoring spotted owl behavior during those premanipulation hours and through unattended video camera coverage at other times confirmed undisturbed, normal activity. While

monitored spotted owls were aware of the cameras, we recorded no unusual behaviors or changes in activity patterns in response to their presence. In fact, spotted owls that used the camera trees, and sometimes the same branch, for perching continued to do so after camera placement. Neither locator balloons, which were above the canopy and obscured by intervening vegetation, nor flashlights with directional beams pointed skyward were normally visible to manipulated spotted owls. Our data collection activities did not seem to affect spotted owl responses to experimental manipulations.

Reproductive Success

Other noise disturbance research suggests aircraft overflights alone have a negligible effect on raptor reproductive success and young fledged per nest (Platt 1977, Anderson et al. 1989, Ellis et al. 1991). We believe the small, nonsignificant decrease in reproductive success between manipulated (n = 17) and nonmanipulated (n = 5) sites in our study was attributable to natural attrition inherent in the larger manipulation sample.

Our ability to detect a biologically meaningful difference in reproductive success and productivity between manipulated and nonmanipulated nests was limited by population size. Sample sizes of 116 nests for measuring success and 362 nests for measuring productivity would have been required to reach an adequate power level of 0.80. With only 30–50 spotted owl territories reproductively active on the Sacramento Ranger District each year, and only about 80% of those sites successfully producing young in a good year, adequate power levels can never be reached.

Although we were not able to relate the number of flushes or the number of manipulations to the number of young fledged, both parameters should be addressed in noise disturbance research (Awbrey and Bowles 1990). Helicopter-induced flushes have been found to affect the number of young gyrfalcons (*Falco rusticolus*) that fledged in Alaska (Platt 1977), while Awbrey and Bowles (1990) hypothesized flushes may be the best predictor of eventual reproductive loss.

Flush Response

The proportion of spotted owls flushing in response to a manipulation was negatively related to stimulus distance and positively related to noise level. Grubb and King (1991), McGarigal et al. (1991), Stalmaster and Kaiser (1997), and others reported similar findings for bald eagles (*Haliaeetus leucocephalus*), with response to human activity increasing as stimulus distance decreased. Platt (1977) describes a comparable pattern for gyrfalcon response to helicopter flights.

The dose-response relations of flush frequency with distance (Fig. 2) and noise level (Fig. 4) indicated that chain saws, although not as loud as helicopters, caused spotted owls to respond from farther away and at higher frequencies. Results for both stimuli are consistent with a model derived from 9 studies of aircraft disturbance effects on several species of nesting raptors (Bowles et al. 1990), which predicts increasing flush probability with increasing noise levels.

Temporal Variation in Spotted Owl Flush Response.—Most studies have not examined the effects of human activities during the incubation and fledgling phases of the nesting season, primarily because of concerns over causing early nesting failure and premature fledging by juveniles (Awbrey and Bowles 1990). However, we observed a strong dichotomy in response behavior between prefledging and postfledging periods, with female spotted owls only flushing after their chicks had left the nest. Spotted owls, like other raptors, appear reluctant to leave the nest during the incubation and nestling phases (Craig and Craig 1984, Fraser et al. 1985, Anderson et al. 1989, Ellis et al. 1991). For bald eagles, flush frequency increased as the nesting season progressed and nest attendance declined, with the highest response rate occurring postfledging (Grubb and Bowerman 1997).

The fact that adult spotted owls were more likely to flush in response to manipulations later in the reproductive cycle also suggests a decrease in adult defensive or protective behavior as juveniles mature. Because adult spotted owls roosted at increasing distances from maturing juveniles, flush distance may become less critical as the fledgling phase progresses, especially because adults did not flush farther than their original distance from juveniles. Although season and nesting phase influence avian response to disturbance (Thiessen 1957, Knight and Temple 1986), prior experience, habituation, and animal temperament may be more important factors (Hart 1985, Manci et al. 1988). In

fact, prior experience may be the best indicator of animal response to overflights (Bowles 1995).

Distance and Sound Thresholds.—Our distance-response threshold for spotted owls was similar to that of most other raptor species exposed to aircraft overflights (N. F. R. Snyder et al. 1978. An evaluation of some potential impacts of the proposed Dade County training jetport on the endangered Everglade Kite, unpublished report. U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Maryland, USA; Craig and Craig 1984, Anderson et al. 1989). For example, Grubb and Bowerman (1997) recommended helicopter survey flights remain >150 m from bald eagle nests and be <1 min in duration. Despite the aggressive nature of our testing regime (i.e., close proximity, repeated exposure, little or no prior experience), spotted owl behavioral responses were minimal when noise disturbance stimuli were >105 m away, and reproduction was not detrimentally affected. Because the 48 RQS varies every flight path during normal training operations over the forest, spotted owls on the LNF would not likely receive as much military helicopter disturbance within any year as the manipulated pairs received during this study.

Distance was a better predictor of spotted owl response to helicopter flights than noise levels. Even when we controlled for distance, noise levels varied among helicopter flights more than among chain saw manipulations. Helicopter noise varied not only with distance but also with rotor pitch, rotor torque, power levels, pilot technique, aircraft loading, speed, topography, and weather. Awbrey and Bowles (1990) described distance as the most commonly used surrogate for noise exposure in the animal effects literature, and suggested distance may be the best representative for the relation between stimulus and response measures. Grubb and King (1991) determined distance was the single most important predictor of bald eagle response in a classification tree model. Their model ranked noise sixth, behind distance, duration of disturbance, visibility, number of disturbances per event, and stimulus position relative to the affected eagle.

Helicopters Versus Chain Saws.—Few researchers have directly compared differences in raptor responsiveness between aerial and ground-based disturbances. In our study, ground-based disturbances elicited a greater flush response than aerial disturbances. Nesting bald eagles in Arizona showed the highest response frequency and severity of response to ground disturbances, followed by aquatic and aerial disturbances (Grubb and King 1991). Awbrey and Bowles (1990), in their meta-analysis of noise disturbance research on raptors, noted aircraft overflights were less detrimental than common ground-based activities such as hiking.

Spotted owls may have perceived helicopters as less threatening than chain saws because of their shorter duration, gradual crescendo in noise levels, minimal visibility, and lack of association with human activity. Helicopters would have elicited greater spotted owl response if exposure times were increased through slower maneuvers such as hovering. Chain saws started abruptly with an associated startle effect, whereas approaching helicopters were always preceded by a gradual increase in noise levels. Disturbing activities in close proximity to a spotted owl's location may also be more visible and therefore elicit a greater response than an activity farther away, regardless of noise level. However, we believe visibility had very limited effect on our results; helicopters were usually not visible, or only briefly so, to spotted owls roosting within or beneath the forest canopy. Grubb and Bowerman (1997) found aircraft visibility had little effect on bald eagle responsiveness. Although chain saws were also operated out of sight of reference spotted owls in all but a few instances, field crews had to set up recording equipment beneath the spotted owls for both types of manipulations. Subsequent chain saw operation may have been associated more with this ground-based human activity. In addition, raptors may be less disturbed by aerial manipulations because of their use of that medium (Gilmer and Stewart 1983).

Alert Response

Spotted owls initially responded to noise disturbances by turning toward the source. This orienting, alert response is an example of an animal's awareness of the disturbance through increased readiness to respond (Archer 1979, Brown 1990). Orienting response becomes progressively less frequent with repeated exposure to the same stimuli (Archer 1979). The relatively quick return to predisturbance behavior we documented is consistent with Ellis (1981), who showed heart rates of prairie falcons (*Falco mexicanus*) exposed to aircraft overflights returned to predisturbance levels within 5 min.

Our mean alert response threshold (403 m) corroborates a regional U.S. Fish and Wildlife Service policy that recommends a 400-m buffer zone around spotted owl nest sites (C. Torez, U.S. Fish and Wildlife Service, personal communication).

Habituation

Habituation is defined as an animal's progressive loss in responsiveness toward a stimulus and is an important determinant in overall response behavior (Peeke and Herz 1973). Although statistically insignificant due to small sample size, experimental spotted owls were less likely to flush (0 of 8) in response to helicopter and chain saw manipulations later in the season than control spotted owls (5 of 8 or 62%), suggesting that spotted owls may have been habituating to manipulation testing. Platt (1977) and Anderson et al. (1989) observed a similar decrease in flush response to aircraft overflights between experienced and relatively naive raptors. In northcentral Michigan, a pair of nesting bald eagles near a military air base was 14% less responsive and the median distance to aircraft eliciting a response was half that (400 m vs. 800 m) of 5 other pairs more remotely located (Grubb et al. 1992). In addition, response duration decreased and time to alert increased during the 1996 field season. However, we consider these trends weaker evidence for habituation because the influence of seasonal factors, such as nesting phase, cannot be differentiated.

Prey Delivery Rates and Related Behaviors

The effects of human activity on a raptor's ability to procure prey during the nesting season has not been well documented (Awbrey and Bowles 1990). Holthuijzen et al. (1990) found that prey delivery rates did not differ between experimental and control sites for prairie falcons exposed to construction and mining activity. Comparable examples for nocturnal raptors are lacking. We found prey delivery rates were highly and positively correlated with stimulus distance. Thus, manipulations in close proximity to spotted owl territories may affect prey delivery rates. The estimated threshold for detrimentally affecting prey deliveries (96 m) indicates a subflushing response consistent with the 105-m flush threshold and emphasizes the potential importance of this threshold distance. However, these threshold findings are based on a specific range of manipulations per territory; changes in stimulus frequency, duration, timing, and distance could strongly influence overall spotted owl response behavior.

Weather conditions should be considered when determining the effects of human activity on raptors (Schueck and Marzluff 1995). To control for possible weather effects, we did not conduct any manipulations during periods of inclement weather. This approach was consistent with the training protocol for the 48 RQS, which limited activity during such periods. However, factors such as precipitation, wind, and clouds can limit foraging ability of raptors (Brown et al. 1988, Bosakowski 1989, Flemming and Smith 1990) and thereby place greater importance on the next available foraging times, when disturbance could become more critical.

MANAGEMENT IMPLICATIONS

This research differs from previous noise disturbance studies by a unique combination of factors: (1) interpretation of noise levels via owl-weighting, which is more specific to the subject animal's hearing sensitivity than the generalized and less applicable A- or flat-weighting; (2) field experimentation with a threatened species in its natural habitat during a normal nesting season; and (3) controlled experimentation with the same resource and military aircraft, personnel, and flight profiles that initially raised the question of potential disturbance. A progressive, incremental, and conservative approach made this experimentation possible with no resultant negative effect on spotted owl activity or productivity.

Potentially detrimental noise levels were our initial and primary concern, but stimulus distance evolved as a more easily defined, quantified, and managed characteristic. Spotted owl response data were readily analyzed by distance because manipulation stimuli were presented by distance. In addition, distance results also directly translate into practical management implications. However, any application of our spotted owl response distances to develop management protocols for spotted owls elsewhere is inherently limited because it is predicated on having the same stimulus in a context similar to our experimentation. Alternatively, considering spotted owl response in terms of noise levels enables our results to be more generally ap-

plied, with due caution, to other types of helicopters under more varying conditions. This distinction is explained by the fact that noise level at the target species is the final measure of stimulus noise, whereas distance is only 1 of numerous intermediate factors (such as terrain, vegetation, atmospheric conditions, stimulus type, size, operation, etc.) that can affect noise level at the target.

Nonetheless, our data indicate the following recommendations for management of helicopter noise near Mexican spotted owls:

(1) At comparable distances, helicopter overflights were less disturbing to spotted owls than chain saws. This result validates, for this species and aircraft, the already established pattern that ground-based activities are typically more disturbing to raptors than aerial activities.

(2) Spotted owls did not flush when helicopter SEL noise levels were ≤102 dBO (92 dBA). Hence, helicopter noise levels below this threshold should not detrimentally affect nesting spotted owls.

(3) A 105-m radius, hemispherical protection zone should eliminate spotted owl flush response to helicopter overflights on the LNF. Zero flush response beyond 105 m for both helicopters and the more disturbing chain saws support this conclusion. Detrimental effects on prey delivery rates should also be minimized because the estimated threshold for this subflushing response (96 m) was <105 m.

(4) Short duration, single pass, single aircraft overflights had little effect on spotted owls. Other flight maneuvers involving circling, hovering, landing, etc., with potential increases in duration, proximity, or noise levels were not included in our experimentation.

(5) Our behavioral data indicate diurnal flights will likely have less potential for disrupting critical spotted owl activity than nocturnal flights. However, during nighttime hours, the 3 hr following sunset and preceding dawn were most important. Helicopter overflights between these nocturnal hours should minimize effects on spotted owl behavior.

(6) Considering the frequency of our manipulation testing, we recommend separating potential owl overflights along the same route by at least 7 days. Because flights over the same sites were separated as much as possible to minimize effects during our testing, data on the potential effects of more frequent overflights are lacking. However, actual rescue training flights avoid using the same route and therefore should not affect the same nest site twice in a breeding season.

(7) Although multiple flights over any 1 site are not recommended, our trend data indicate the likelihood of habituation with repeated exposures and as the nesting season progressed. Thus, naive, unexposed spotted owls may be more affected than spotted owls that have previously experienced overflights.

(8) Spotted owl flush response to helicopter overflights did not differ between the nesting and nonnesting seasons. Within the context of our experimentation, we found no substantive evidence that helicopter overflights during the nesting season detrimentally affected spotted owl success or productivity.

In conclusion, these research findings are specific to Mexican spotted owls and Pave Hawk helicopters, as well as to the seasons and habitat associated with our testing. Therefore, extrapolation to different avian genera or species, or other aircraft and locations, must be done with caution. For example, changes in forest type or elevation alone may influence prey availability and delivery rates, which may in turn influence spotted owl response behavior. We also caution against use of these findings to infer how spotted owls would respond under different circumstances that were not directly tested, such as spotted owl responses during early courtship and incubation, responses to >1 helicopter or overflight, or responses in different nesting habitat or under different foraging conditions. While our research was effective in answering the original, specific disturbance question, these results must be qualified by the limiting context of their derivation when applied to broader managerial questions.

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DEPARTMENT OF ENERGY & CLIMATE CHANGE

Overarching National Policy Statement for Energy (EN-1)

Planning for new energy infrastructure

July 2011

Department of Energy and Climate Change

Overarching National Policy Statement for Energy (EN-1)

Presented to Parliament pursuant to Section 5(9)
of the Planning Act 2008

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4.2 Environmental Statement

4.2.1 All proposals for projects that are subject to the European Environmental Impact Assessment Directive[74] must be accompanied by an Environmental Statement (ES) describing the aspects of the environment likely to be significantly affected by the project[75]. The Directive specifically refers to effects on human beings[76], fauna and flora, soil, water, air, climate, the landscape, material assets and cultural heritage, and the interaction between them. The Directive requires an assessment of the likely significant effects of the proposed project on the environment, covering the direct effects and any indirect, secondary, cumulative, short, medium and long-term, permanent and temporary, positive and negative effects at all stages of the project, and also of the measures envisaged for avoiding or mitigating significant advers effects.

4.2.2 To consider the potential effects, including benefits, of a proposal for a project, the IPC will find it helpful if the applicant sets out information on the likely significant social and economic effects of the development, and shows how any likely significant negative effects would be avoided or mitigated. This information could include matters such as employment, equality, community cohesion and well-being.

4.2.3 For the purposes of this NPS and the technology-specific NPSs the ES should cover the environmental, social and economic effects arising from pre-construction, construction, operation and decommissioning of the project. In some circumstances (for example, gas pipe-lines) it may be appropriate to assess effects arising from commissioning infrastructure once it is completed but before it comes into operation. Details of this and any other additional assessments are set out where necessary in sections on individual impacts in this NPS and in the technology-specific NPSs. In the absence of any additional information on additional assessments, the principles set out in this Section will apply to all assessments.

4.2.4 When considering a proposal the IPC should satisfy itself that likely significant effects, including any significant residual effects taking account of any proposed mitigation measures or any adverse effects of those measures, have been adequately assessed. In doing so the IPC should also examine whether the assessment distinguishes between the project stages and identifies any mitigation measures at those stages. The IPC should request further information where necessary to ensure compliance with the EIA Directive.

74 Council Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment, amended by Directives 97/11/EC and 2003/35/EC. In respect of energy NSIPs, Annex 1 of the directive applies to thermal power stations, nuclear power stations, waste-disposal installations for the incineration, chemical treatment or land fill of toxic and dangerous wastes. Under Annex 2 it applies to industrial installations for the production of electricity, steam and hot water (i.e. CHP), industrial installations for carrying gas, steam and hot water; transmission of electrical energy by overhead cables, surface storage of natural gas, underground storage of combustible gases and installations for hydroelectric energy production.

75 The Infrastructure Planning (Environmental Impact Assessment) Regulations 2009 (SI 2009/2263).

76 The effects on human beings includes effects on health.

4.2.5 When considering cumulative effects, the ES should provide information on how the effects of the applicant's proposal would combine and interact with the effects of other development (including projects for which consent has been sought or granted, as well as those already in existence)[77]. The IPC may also have other evidence before it, for example from appraisals of sustainability of relevant NPSs or development plans, on such effects and potential interactions. Any such information may assist the IPC in reaching decisions on proposals and on mitigation measures that may be required.

4.2.6 The IPC should consider how the accumulation of, and interrelationship between, effects might affect the environment, economy or community as a whole, even though they may be acceptable when considered on an individual basis with mitigation measures in place.

4.2.7 In some instances it may not be possible at the time of the application for development consent for all aspects of the proposal to have been settled in precise detail. Where this is the case, the applicant should explain in its application which elements of the proposal have yet to be finalised, and the reasons why this is the case.

4.2.8 Where some details are still to be finalised the ES should set out, to the best of the applicant's knowledge, what the maximum extent of the proposed development may be in terms of site and plant specifications, and assess, on that basis, the effects which the project could have to ensure that the impacts of the project as it may be constructed have been properly assessed[78].

4.2.9 Should the IPC determine to grant development consent for an application where details are still to be finalised, it will need to reflect this in appropriate development consent requirements. Clearly, if development consent is granted for a proposal and at a later stage the developer wishes for technical or commercial reasons to construct it in such a way that its extent will be greater than has been provided for in the terms of the consent, it may be necessary to apply for a change to be made to the development consent, and the application to change the consent may need to be accompanied by further environmental information to supplement the original ES.

4.2.10 To help the IPC consider thoroughly the potential effects of a proposed project in cases where the EIA Directive does not apply and an ES is not therefore required, the applicant should instead provide information proportionate to the scale of the project on the likely significant environmental, social and economic effects. References to an Environmental Statement in this NPS should be taken as including a statement which provides this information, even if the EIA Directive does not apply.

77 For guidance on the assessment of cumulative effects, see, for example, Circular 02/99, Environmental impact assessment, or *Guidelines for the Assessment of Indirect and Cumulative Impacts as well as Impact Interactions* (http://ec.europa.eu/environment/eia/eia-studies-and-reports/guidel.pdf).

78 Case law (for example Rochdale MBC Ex. Parte C Tew 1999) provides a legal principle that indicative sketches and layouts cannot provide the basis for determining applications for EIA development. The "Rochdale Envelope" is a series of maximum extents of a project for which the significant effects are established. The detailed design of the project can then vary within this 'envelope' without rendering the ES inadequate.

4.2.11 In this NPS and the technology-specific NPSs, the terms 'effects', 'impacts' or 'benefits' should be understood to mean likely significant effects, impacts or benefits.

4.3 Habitats and Species Regulations

4.3.1 Prior to granting a development consent order, the IPC must, under the Habitats and Species Regulations[79], (which implement the relevant parts of the Habitats Directive and the Birds Directive[80] in England and Wales) consider whether the project may have a significant effect on a European site, or on any site to which the same protection is applied as a matter of policy, either alone or in combination with other plans or projects. Further information on the requirements of the Habitats and Species Regulations can be found in a Government Circular[81]. Applicants should also refer to Section 5.3 of this NPS on biodiversity and geological conservation. The applicant should seek the advice of Natural England and/or the Countryside Council for Wales, and provide the IPC with such information as it may reasonably require to determine whether an Appropriate Assessment is required. In the event that an Appropriate Assessment is required, the applicant must provide the IPC with such information as may reasonably be required to enable it to conduct the Appropriate Assessment. This should include information on any mitigation measures that are proposed to minimise or avoid likely effects.

4.4 Alternatives

4.4.1 As in any planning case, the relevance or otherwise to the decision-making process of the existence (or alleged existence) of alternatives to the proposed development is in the first instance a matter of law, detailed guidance on which falls outside the scope of this NPS. From a policy perspective this NPS does not contain any general requirement to consider alternatives or to establish whether the proposed project represents the best option.

4.4.2 However:

- applicants are obliged to include in their ES, as a matter of fact, information about the main alternatives they have studied. This should include an indication of the main reasons for the applicant's choice, taking into account the environmental, social and economic effects and including, where relevant, technical and commercial feasibility;
- in some circumstances there are specific legislative requirements, notably under the Habitats Directive, for the IPC to consider alternatives. These should also be identified in the ES by the applicant; and

79 The Conservation of Habitats and Species Regulations 2010 (SI2010/490).
80 Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora; Council Directive 2009/147/EC on the conservation of wild birds.
81 Government Circular: Biodiversity and Geological Conservation – Statutory Obligations and their impact within the Planning System (ODPM 06/2005, Defra 01/2005) available via TSO website www.tso.co.uk/bookshop. It should be noted that this document does not cover more recent legislative requirements. Where this circular has been superseded, reference should be made to the latest successor document.

5.3 Biodiversity and geological conservation

Introduction

5.3.1 Biodiversity is the variety of life in all its forms and encompasses all species of plants and animals and the complex ecosystems of which they are a part. Geological conservation relates to the sites that are designated for their geology and/or their geomorphological importance.

5.3.2 The wide range of legislative provisions at the international and national level that can impact on planning decisions affecting biodiversity and geological conservation issues are set out in a Government Circular[97]. A separate guide sets out good practice in England in relation to planning for biodiversity and geological conservation[98].

Applicant's assessment

5.3.3 Where the development is subject to EIA the applicant should ensure that the ES clearly sets out any effects on internationally, nationally and locally designated sites of ecological or geological conservation importance, on protected species and on habitats and other species identified as being of principal importance for the conservation of biodiversity. The applicant should provide environmental information proportionate to the infrastructure where EIA is not required to help the IPC consider thoroughly the potential effects of a proposed project.

5.3.4 The applicant should show how the project has taken advantage of opportunities to conserve and enhance biodiversity and geological conservation interests.

IPC decision making

5.3.5 The Government's biodiversity strategy is set out in 'Working with the grain of nature'[99]. Its aim is to ensure:

- a halting, and if possible a reversal, of declines in priority habitats and species, with wild species and habitats as part of healthy, functioning ecosystems; and
- the general acceptance of biodiversity's essential role in enhancing the quality of life, with its conservation becoming a natural consideration in all relevant public, private and non-governmental decisions and policies.

5.3.6 In having regard to the aim of the Government's biodiversity strategy the IPC should take account of the context of the challenge of climate change: failure to address this challenge will result in significant adverse impacts to biodiversity. The policy set out in the following sections recognises the need

97 Government Circular: Biodiversity and Geological Conservation – Statutory Obligations and their Impact within the Planning System (ODPM 06/2005, Defra 01/2005) available via TSO website www.tso.co.uk/bookshop. It should be noted that this document does not cover more recent legislative requirements, such as the Marine Strategy Framework Directive.

98 Planning for Biodiversity and Geological Conservation: A Guide to Good Practice (March 2006).

99 'Working with the grain of nature' applies in England only.

to protect the most important biodiversity and geological conservation interests. The benefits of nationally significant low carbon energy infrastructure development may include benefits for biodiversity and geological conservation interests and these benefits may outweigh harm to these interests. The IPC may take account of any such net benefit in cases where it can be demonstrated.

5.3.7 As a general principle, and subject to the specific policies below, development should aim to avoid significant harm to biodiversity and geological conservation interests, including through mitigation and consideration of reasonable alternatives (as set out in Section 4.4 above); where significant harm cannot be avoided, then appropriate compensation measures should be sought.

5.3.8 In taking decisions, the IPC should ensure that appropriate weight is attached to designated sites of international, national and local importance; protected species; habitats and other species of principal importance for the conservation of biodiversity; and to biodiversity and geological interests within the wider environment.

International Sites

5.3.9 The most important sites for biodiversity are those identified through international conventions and European Directives. The Habitats Regulations provide statutory protection for these sites but do not provide statutory protection for potential Special Protection Areas (pSPAs) before they have been classified as a Special Protection Area. For the purposes of considering development proposals affecting them, as a matter of policy the Government wishes pSPAs to be considered in the same way as if they had already been classified. Listed Ramsar sites should, also as a matter of policy, receive the same protection[100].

Sites of Special Scientific Interest (SSSIs)

5.3.10 Many SSSIs are also designated as sites of international importance and will be protected accordingly. Those that are not, or those features of SSSIs not covered by an international designation, should be given a high degree of protection. All National Nature Reserves are notified as SSSIs.

5.3.11 Where a proposed development on land within or outside an SSSI is likely to have an adverse effect on an SSSI (either individually or in combination with other developments), development consent should not normally be granted. Where an adverse effect, after mitigation, on the site's notified special interest features is likely, an exception should only be made where the benefits (including need) of the development at this site[101], clearly outweigh both the impacts that it is likely to have on the features of the site that make it of special scientific interest and any broader impacts on the national network of SSSIs. The IPC should use requirements and/or planning

100 See http://www.jncc.gov.uk/page-161

101 'At this site' applies the language in PPS9: *Biodiversity and Geological Conservation*. The benefits of the development 'at this site' should be interpreted as including any benefits which are not dependent on a particular location.

obligations to mitigate the harmful[102] aspects of the development and, where possible, to ensure the conservation and enhancement of the site's biodiversity or geological interest.

Marine Conservation Zones

5.3.12 Marine Conservation Zones (MCZs) (Marine Protected Areas in Scotland), introduced under the Marine and Coastal Access Act 2009, are areas that have been designated for the purpose of conserving marine flora or fauna, marine habitats or types of marine habitat or features of geological or geomorphological interest. The protected feature or features and the conservation objectives for the MCZ are stated in the designation order for the MCZ, which provides statutory protection for these areas implemented by the MMO (see paragraph 1.2.2). As a public authority, the IPC is bound by the duties in relation to MCZs imposed by sections 125 and 126 of the Marine and Coastal Access Act 2009.

Regional and Local Sites

5.3.13 Sites of regional and local biodiversity and geological interest, which include Regionally Important Geological Sites, Local Nature Reserves and Local Sites, have a fundamental role to play in meeting overall national biodiversity targets; contributing to the quality of life and the well-being of the community; and in supporting research and education. The IPC should give due consideration to such regional or local designations. However, given the need for new infrastructure, these designations should not be used in themselves to refuse development consent.

Ancient Woodland and Veteran Trees

5.3.14 Ancient woodland is a valuable biodiversity resource both for its diversity of species and for its longevity as woodland. Once lost it cannot be recreated. The IPC should not grant development consent for any development that would result in its loss or deterioration unless the benefits (including need) of the development, in that location[103] outweigh the loss of the woodland habitat. Aged or 'veteran' trees found outside ancient woodland are also particularly valuable for biodiversity and their loss should be avoided[104]. Where such trees would be affected by development proposals the applicant should set out proposals for their conservation or, where their loss is unavoidable, the reasons why.

102 In line with the principle in paragraph 4.2.11, the term 'harm' should be understood to mean 'significant harm'.

103 "in that location" applies the language in PPS9: *Biodiversity and Geological Conservation*. The benefits of the development in that location should be interpreted as including any benefits which are not dependent on a particular location.

104 This does not prevent the loss of such trees where the IPC is satisfied that their loss is unavoidable.

DEPARTMENT OF ENERGY & CLIMATE CHANGE

National Policy Statement for Nuclear Power Generation (EN-6)

Volume II of II - Annexes

Planning for new energy infrastructure

July 2011

Department of Energy and Climate Change

National Policy Statement for Nuclear Power Generation (EN-6)

Volume II of II – Annexes

Presented to Parliament pursuant to section 5(9) of the Planning Act 2008.

July 2011

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of IROPI and the development and implementation of compensatory measures in line with the strategic measures set out below. The Government's findings in respect of Article 6(4) of the Habitats Directive and this NPS do not automatically transfer directly to individual projects and the Nuclear NPS does not in any way reduce the duty on the IPC to fulfil the legal requirements of the Habitats Directive.

A.7 Compensatory Measures

A.7.1 Article 6(4) of the Habitats Directive (Regulation 105 of the Habitats Regulations[18]) requires that where, in spite of a negative assessment on a Natura 2000 site(s) integrity, the competent authority proceeds with the plan on the basis of IROPI, any necessary compensatory measures are taken to ensure that the overall coherence of the Natura 2000 network is protected.

A.7.2 Given the strategic nature of the Habitats Regulations Assessment process for this NPS, the inherent uncertainties of the Appropriate Assessment's conclusions and the potential changes that may occur as the plan is implemented[19], it is not possible at this stage to specify the precise nature or location of any compensatory measures that might be required.

A.7.3 The role of the plan is, therefore, to provide a robust framework through the direction it provides to the IPC that sets out the broad parameters for compensation measures, should they be required following the more detailed site level assessments undertaken for plan implementation.

A.7.4 All project level Habitats Regulations Assessments must take account of the potential adverse effects and the proposed avoidance and mitigation measures identified through the strategic level assessment(s).

A.7.5 Where site level assessments identify that compensation is required it must meet the following criteria and be:

- appropriate for the area and the loss caused by the project;
- capable of protecting the overall coherence of the Natura 2000 network;
- capable of implementation;

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18 The Conservation of Habitats and Species Regulations 2010 (SI2010/490)

19 The Habitats Regulations Assessment of the NPS has noted that avoidance and mitigation measures proposed by the assessment may minimise effects (to the point where integrity is no longer affected) or cancel out the negative impacts predicted such that the site level developments may proceed without the need to meet additional requirements under the Habitats Directive. This will need to be assessed on a case by case basis.

- capable of ensuring that the Natura 2000 site is not irreversibly affected by the project before the compensation is in place;
- directed in measurable proportions to the habitats and species negatively affected;
- related to the same biogeographical region (within the UK);
- serves functions that are comparable to those that motivated the original area's submission for designation; and
- clearly defined, with implementation goals and managed so that the compensatory measures can achieve the goal of maintaining or improving the overall coherence of the Natura 2000 network.

the potential effects of climate change, storm surge and tsunami. This takes into account the potential identified by the Environment Agency to protect the site and to mitigate risks although, as with all sites, the potential effects of any mitigation on the surrounding area will have to be carefully considered as part of a flood risk assessment should any application be forthcoming.

Policy notes

C.2.43 See the relevant guidance in EN-1, including that on flood risk and climate change adaptation.

C.2.44 See the relevant guidance in Part 3 of this NPS, including that on flood risk.

D2: Coastal processes

Analysis

C.2.45 The Environment Agency has advised that development at the site could potentially avoid or mitigate the effects of coastal erosion or other landscape change scenarios throughout its operational lifetime, including the potential effects of climate change[51]. Final proposals are likely to require mitigating actions, for instance the Environment Agency has advised that any new development at Bradwell could be built on higher or raised ground. The Environment Agency has also noted that positioning of the site could limit any future long term effects of coastal erosion.

C.2.46 The Appraisal of Sustainability has identified possible secondary impacts on coastal processes, hydrodynamics and sediment transport from any necessary new or upgraded coastal defences. It has found that mitigation may be possible through appropriate design and construction of defences. The Environment Agency has also advised that the impacts of a power station on the evolution and geomorphology of the Estuary channel may need to be considered, including the impact of elements such as outfalls of cooling water on the adjacent areas in the Estuary and what potential for change this may cause in estuarine flows.

C.2.47 Responses were concerned about the wider impact of any coastal defences required at the site. These concerns centred on coastal processes and squeeze on two internationally designated sites, the Dengie Special Protection Area (SPA) and Ramsar site and Essex Estuaries Special Area of Conservation (SAC) and mitigation was not thought to be likely. The forthcoming Shore Line Management Plan for Essex was also referenced, it was stated that as this will only consider a period up to 2100,

51 See entry D1 in the table “The SSA criteria and how the sites were assessed” at the end of this Annex for details on the potential lifetime of the site and the period this assessment covers.

more detailed modelling and scenario building for the next 200 years was required

C.2.48 The Environment Agency has noted for all nominated sites that protecting the site from flood risk now and in the future prevents the coastline and Estuary from changing and adapting naturally. The Habitats Regulations Assessment report for Bradwell identified that development, particularly proposals for upgraded coastal protection and a marine landing facility, would encroach directly on the margins of the Essex Estuaries SAC and the Blackwater Estuary SPA/Ramsar and the Dengie Estuary SPA/Ramsar sites[52]. These designations are currently under threat from the effects of coastal squeeze which has been identified as a problem in the area. However, the extent of the loss and/or fragmentation of marine, intertidal and terrestrial habitats likely to be attributable to the construction of nuclear reactors, construction areas and other infrastructure and facilities relating to the operation of the nuclear power station is currently unknown. This is because the project design and exact scope of the development and the requirements for coastal or sea defence infrastructure remain undetermined at this stage.

C.2.49 The Habitats Regulations Assessment site report has set out a number of suggested avoidance and mitigation measures. This could include avoiding or minimising losses of habitat through sensitively designed sea defences such as soft engineering for any upgraded coastal protection. It will ultimately be the responsibility of the nominator to suggest appropriate mitigation measures and these would be assessed at the project level.

C.2.50 The second generation of Shoreline Management Plans (SMP2) are designed to provide a 'route map' for local authorities and other decision makers to move from the present situation towards meeting future needs of the coastline. SMP2s identify the most sustainable approaches to managing the risks to the coast in the short term (0-20 years), medium term (20-50 years) and long term (50-100 years). Within these timeframes, SMP2s also include an action plan that prioritises what work is needed to manage coastal processes into the future, and where it will happen.

Assessment

C.2.51 The site passes this criterion. Based on the advice above, it is reasonable to conclude at a strategic level that a nuclear power station within the site could be protected against coastal erosion and other landscape change scenarios, including the potential effects of climate change, for the lifetime of the station, taking into account countermeasures and mitigating actions.

52 DECC, *Habitats Regulations Assessment: site report for Bradwell*, October 2010, www.energynpsconsultation.decc.gov.uk

The potential impacts of development on designated habitats will be taken into account in the project level assessments (including a further project level Habitats Regulations Assessment and an Environmental Statement reporting the findings of a detailed Environmental Impact Assessment) and considered by the IPC as part of the application for development consent.

Policy notes

C.2.52 See Section 5.5 of EN-1 and Section 3.9 of this NPS on coastal change.

D3: Proximity to hazardous industrial facilities and operations

Analysis

C.2.53 A small area in the south west tip of the site is within the land use planning consultation zones for the former COMAH[53] establishment at Supergas. This was decommissioned in 1999. The land may still be covered by a Hazardous Substances Consent which is administered by the local planning authority. However, the map at the end of this annex illustrates that the area of overlap is only an extremely small portion of the nominated land.

C.2.54 The Office for Nuclear Regulation has advised that it is reasonable to conclude that the site can be protected against the risk arising from proximity to hazardous facilities throughout its lifetime taking into account possible countermeasures and mitigating actions. Given the small area of overlap with the consultation zones for the extant Supergas facility, it is likely that should a hazard still be posed, mitigating actions would be available such as siting key buildings away from that area. The Office for Nuclear Regulation has advised that as with all sites during licensing the licence applicant to the Office for Nuclear Regulation will also need to take account of the need for countermeasures to protect nuclear operations from any hazards and risks from any nearby notified major hazard pipelines, based on information from the relevant pipeline operators about their routes and fluids being conveyed.

Assessment

C.2.55 The proximity of the extant Supergas facility does not affect the potential suitability of the site at this stage given the scope for avoidance or mitigation if necessary. This site passes this criterion. It is reasonable to conclude that a new nuclear power station at the site could be protected against risk arising from proximity to hazardous facilities throughout its lifetime.

53 Under the Planning (Control of Major Accident Hazards) Regulations 1999. For more information see: http://www.hse.gov.uk/comah/background/comah99.htm#

D7: Nationally designated sites of ecological importance

Analysis

C.8.60 Some responses focused on designated sites including Sizewell Marshes Site of Special Scientific Interest (SSSI) and Leiston-Aldeburgh SSSI, and potential effects on Minsmere-Walberswick Heaths and Marshes SSSI, from which the site boundary includes some land-take. Some responses questioned how direct land take could be mitigated. Responses were particularly concerned that an access road which is reflected in the site boundary could result in the loss of woodland and heathland habitat at Kenton Hills, Goose Hills and Sizewell Belts. Some responses noted that planning permission had been refused in this area in the past.

C.8.61 The Appraisal of Sustainability identified the potential for adverse effects on sites and species considered to be of national nature conservation importance means that significant strategic effects on biodiversity cannot be ruled out at this stage of the appraisal. The Appraisal of Sustainability identifies that there could be potential significant effects at the following SSSIs which are within 5km of the site: Sizewell Marshes SSSI; Minsmere-Walberswick Heaths and Marshes SSSI; Leiston-Aldeburgh SSSI; Alde-Ore Estuary SSSI. The Appraisal of Sustainability also notes that the above designated sites include RSPB reserves adjacent to the site (Minsmere) and within 1.5km to the north (North Warren).

C.8.62 As the site boundary also indicates land-take from Sizewell Marshes SSSI, the Appraisal of Sustainability finds that construction and the presence of development are likely to lead to direct loss and fragmentation of habitats within the Sizewell Marshes SSSI. Sizewell Marshes SSSI is an area of grazing marsh with important assemblages of invertebrates and breeding and winter bird populations.

C.8.63 The Appraisal of Sustainability identified the potential for the mitigation of biodiversity effects on sites of UK wide conservation importance (Sizewell Marshes SSSI), including the creation of replacement habitat. The Appraisal of Sustainability notes that developers could avoid or minimise losses and disturbance to protected species through careful site layout, design, routing, location of the development, associated infrastructure, and construction management and timings. The Appraisal of Sustainability finds that there is potential for habitat creation within the wider area in order to replace lost ‘wet meadows’ habitats of the Sizewell Marshes SSSI, but also finds that it may not be possible to fully compensate for losses of this habitat. The applicant will need to develop an ecological mitigation and management plan to minimise the impacts.

Assessment

C.8.64 The Government notes that the Appraisal of Sustainability has identified potential impacts on nationally designated sites of ecological importance

which it considers of strategic significance. Given the scope for mitigation of biodiversity effects identified in the Appraisal of Sustainability for sites of national importance it is reasonable to conclude that it may be possible to avoid or mitigate impacts to an extent. However, the Appraisal of Sustainability has highlighted that the site includes land take from Sizewell Marshes SSSI that could lead to direct impacts.

C.8.65 The Government has carefully considered whether this site meets this criterion given the direct impact on Sizewell Marshes SSSI. However, given the need to ensure sufficient sites are available for development to meet the Government's energy policy objectives (as described in Part 2 of this NPS), the Government believes that it does. In view of the need for sites and the limited number of potentially suitable sites, the Government does not think the issues in relation to this criterion are sufficient to justify not including the site in this NPS. The Government has also noted that there will be further assessment of any proposal for the site at project level and that EN-1 sets out detailed consideration that must be given to issues related to nationally designated sites, should an application for development consent come forward.

Policy notes

C.8.66 See the relevant guidance in EN-1, including that on the Environmental Statement and biodiversity and geological conservation. See the relevant guidance in Part 3 of this NPS, including that on biodiversity and geological conservation.

C.8.67 The IPC should also refer to the Appraisal of Sustainability for Sizewell and consider whether the applicant's proposals have sufficiently taken into account the issues identified, where they are still relevant.

D8: Areas of amenity, cultural heritage and landscape value

Analysis

C.8.68 The site is entirely within the Suffolk Coast and Heaths AONB. Responses raised concerns about the adverse effects on the AONB particularly in regard to the impact development of an access road to the site could have on the Kenton and Goose Hills and the AONB. In particular responses were concerned that an access road could run through the whole of the AONB and were concerned about the visual impact that this would have on the character of the area.

C.8.69 The SSA has not assessed in detail proposals for associated works such as access roads. Such details could change without affecting the overall strategic suitability of the site. The Government believes that this type of proposal is more appropriately considered by the IPC. There is no presumption that development will take place in the area of the access road. The IPC will need to consider detailed plans using the guidance

www.defra.gov.uk

Biodiversity Offsetting Pilots

Technical Paper: the metric for the biodiversity offsetting pilot in England

March 2012

NATURAL ENGLAND

defra
Department for Environment Food and Rural Affairs

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Habitat hectares assesses these various attributes against 'benchmarks' representing the average characteristics of mature stands of native vegetation of the same community type in a 'natural' or 'undisturbed' condition.

13. The habitat hectares system used in Australia is intensive in terms of the input required to assess the habitat. It requires trained operators to ensure the required levels of consistency. Consultants have to pass an exam before they are allowed to submit assessments (Cara Reece pers. com.).

14. In the USA, where offsets schemes have been running for 30 years, there are a broad range of metrics in use. The majority of assessments in offsetting schemes in the USA make use of an area measurement and a multiplier, and sometimes an approximate quality assessment based on expert opinion (Briggs et al. 2009, BOPP 2009).

Metrics for biodiversity offsetting in England

15. Biodiversity offsetting, where conservation activities are designed to deliver biodiversity benefits in compensation for losses, in a measurable way, are one way of providing compensation for biodiversity loss. We believe that a consistent framework for biodiversity offsetting has the potential to improve the implementation of planning policy requirements for biodiversity compensation.

16. Applying biodiversity offsetting in this way in England would be a new and innovative approach, and there are many aspects which we don't fully understand yet. There are a number of issues that need to be clarified before we decide exactly whether, and how, we can make best use of biodiversity offsetting in England. That is why we are working with local authorities and other partners to test biodiversity offsetting in 6 pilot areas.

17. The principles we have used to develop an approach to biodiversity offsetting in England are set out in the Guiding Principles for Biodiversity Offsetting Document, available on Defra's website. The principles include the following:

Offsetting should:

- not change existing levels of protection for biodiversity
- expand and restore habitats, not merely protect the extent and condition of what is already there
- contribute to enhancing England's ecological network by creating more, bigger, better and joined areas for biodiversity (as discussed in *Making Space for Nature*[3])
- be managed at the local level as far as possible

[3] http://archive.defra.gov.uk/environment/biodiversity/documents/201009space-for-nature.pdf

- be as simple and straightforward as possible, for developers, local authorities and others

- be transparent, giving clarity on how the offset calculations are derived and allowing people to see how offset resources are being used.

18. The approach to the metric we will use for the biodiversity offsetting pilot is described in these papers, and reflects these principles.

19. The proposed system is a variation of the habitat hectares approach and draws heavily on the work done by Treweek et al. for the Defra scoping study on offsets (2009).

Habitat type

20. The metric we propose for the biodiversity offsetting pilots is based on habitats. Development sites need to be mapped and divided into habitat parcels. The offset requirement can then be worked out on a habitat basis. The same basic approach can then be used to work out what level of compensation an offset project is able to deliver.

21. Habitats are pre-assigned to one of three habitat type bands (**Figure 1** below). Habitats are assigned to these bands on the basis of their distinctiveness. Distinctiveness includes parameters such as species richness, diversity, rarity (at local, regional, national and international scales) and the degree to which a habitat supports species rarely found in other habitats (Treweek et al 2010). Details of the distinctiveness bands can be found in **Appendix 1 - Distinctiveness Bands for the Biodiversity Offsetting Pilot**, which is available on Defra's website.

22. One of the guiding principles for developing our approach to offsetting is that it should result in an improvement in the extent or condition of the ecological network. To do this the focus of habitat restoration or creation through offsetting should be on priority habitats[4]. Where development is taking place on habitats in the low distinctiveness band, the offset actions should result in expansion or restoration of habitats in the medium or, preferably, high distinctiveness band. At no time should an offset result in "trading down", for instance in the replacement of habitat of high distinctiveness with creation or restoration of a habitat of medium distinctiveness. Habitats that are of high distinctiveness would generally be expected to be offset with "like for like" i.e. the compensation should involve the same habitat as was lost.

23. Some very valuable habitats are either very rare, difficult/impossible to recreate, or both. Whilst development on these habitats would be unlikely, if a local planning authority did decide that a development should go ahead on this type of habitat, any compensation

[4] Section 41(S41) of the Natural Environment and Rural Communities (NERC) Act requires the Secretary of State to publish a list of habitats and species which are of principal importance for the conservation of biodiversity in England. Further information about this list can be found here: http://www.naturalengland.org.uk/ourwork/conservation/biodiversity/protectandmanage/habsandspeciesimportance.aspx

would have to be bespoke, and managed on a case by case basis. It would be for the local planning authority to decide if the offsetting mechanism could be used.

24. **Figure 1** shows the habitat bands we are using for the biodiversity offsetting pilots.

Figure 1: Habitat type bands

Habitat type band	Distinctiveness	Broad habitat type covered	Type of offset
High	High	Priority habitat, as defined in Section 41 of the NERC Act[5]	Same band type, and ideally like for like
Medium	Medium	Semi natural	Within band type or trade up
Low	Low	E.g. Intensive agricultural– but may still form an important part of the ecological network in an area.	Trade up

25. As per the guiding principles, decisions about exactly where offsets should be targeted geographically, and towards which conservation priorities, should be taken at the local level as far as possible. In line with this principle, local authorities in pilot areas, working with their partners, could decide to add conditions to the metric to reflect their particular circumstances and priorities, as part of the development of their offsetting strategy. For example, they may decide that a particular habitat is especially important in their area, and therefore would like any offsets provided to compensate for loss of that particular habitat to comprise expansion or restoration of that habitat. They may decide that in their area a particular habitat is in a higher distinctiveness band than that suggested by the national guidance. Where changes to the standard approach are made, the rationale would have to be clearly set out, and the information about the difference available to all potential participants, at the start of the process.

5 http://www.naturalengland.org.uk/ourwork/conservation/biodiversity/protectandmanage/habsandspeciesimportance.aspx

26. In the pilot we want to learn more about how often local authorities want to make these kinds of adjustments, and how it might work in practice.

27. Each band of habitat distinctiveness has a number associated with it as in **Figure 2** below. This is the starting point for calculating the number of "units" of biodiversity per hectare a particular habitat is worth.

Figure 2: Habitat distinctiveness[6]

Habitat distinctiveness	
High	6
Medium	4
Low	2

Habitat condition

28. Different sites and habitats will be in different conditions when they are lost to development, and in addition, offsetting projects will not always involve taking a habitat in poor condition and improving it to good condition. We therefore propose that the metric we use for the biodiversity offsetting pilots takes account of habitat condition, as well as habitat distinctiveness.

29. Condition assessment requires that we have agreed standards and a related methodology for measuring habitat condition. There is currently no standard habitat condition assessment tool, although various methods are used for specific purposes.

30. Perhaps the best known condition assessment tool is Common Standards Monitoring (CSM). This methodology has been devised specifically for monitoring Sites of Special Scientific Interest (SSSIs) and as such it is (a) based around a subset of the most important habitats and (b) is designed to give a very specific output – namely to answer the question "is the site in favourable condition?". It categorises sites into 'favourable', 'favourable recovering' and 'unfavourable'.

31. Whilst suitable for assessing SSSIs, these categories would not work well in the metric described here. They are not evenly spread, and there can be a very wide range within the favourable recovering category. In addition, they may describe the management of the site, rather than the actual condition of the habitat.

[6] [6] Based on the paper "Biodiversity Offsets", Treweek et al.

Insurance

58. A further approach to managing risks is insurance. An offset provider could take out insurance against their failure to deliver the right number of units, in addition to, or instead of, using multipliers.

59. Financial insurance would provide a source of funds for re-attempting the offset project that had failed, thus still allowing the offset provider to meet their obligation in terms of units of biodiversity. The insurance premiums paid by offset providers would likely reflect the type of habitat creation/restoration scheme being undertaken, and therefore its specific risk of failure. In **Appendix 1** to this document, habitats have been assigned to broad risk categories both for expansion (recreation) and restoration.

60. The pilot will help us to learn more about how offset providers choose to manage their risks.

Multipliers and time

61. In delivering offsets there may be a mismatch in the timing of impact and offset, i.e. the difference in time between the negative impact on biodiversity and the offset reaching the required quality or level of maturity, which results in loss of biodiversity for a period of time.

62. This issue could be managed by encouraging the creation of offsets ahead of the impact taking place, either though the setting up of habitat banks or, for projects with a long lead in, by starting the offset work well ahead of the development.

63. However, particularly in the early stages of introducing a new approach to offsetting, many offsets are likely to be developed concurrently with the impact taking place. This will be the case in the biodiversity offsetting pilots. Even where the offset has been started in advance, the time taken for habitats to mature means that there will almost inevitably be a time lag. Where a time lag does occur, a multiplier can be applied to take account of it.

64. Discounting over time is an economic technique used to compare costs and benefits that occur in different time periods based around the principle that, generally, people prefer to receive goods and services now rather than later (more details on discounting can be found in the Treasury Green Book Guidance[12]). Whilst for individuals the evidence for a preference to consume today is good, the evidence as to why society should do this, the ecological basis for it is more complex (for discussion see Annex 5 REMEDE 2008, NOAA 2006).

[12] http://www.hm-treasury.gov.uk/data_greenbook_index.htm

65. Discussions with stakeholders indicate that they support the use of a multiplier to account for the temporal risk in the approach to offsetting we use in England. This is because it would:

- incentivise habitat banking: if the habitat is established there is no need to apply multipliers to manage delivery risks, and to take account of time differences. So more units will be available from a particular area of land.

- create a disincentive for damaging habitats that take a long time to recreate or restore (i.e. many habitats in the 'high' distinctiveness band), by increasing the area of offset needed to compensate for the loss.

66. Where time discounting is used in offset or compensation schemes, for instance in the US and in Defra's Environmental Liability Directive guidance, they tend to use a standard discount rate, for example 7% or 3%, discussed in NOAA 2006 and 3.5%, Defra, 2009. In England, the Treasury Green Book recommends a discount rate of 3.5% to reflect the value society attaches to 'consumption' (i.e. enjoyment of goods and services) at different points in time. It is therefore recommended that this is the rate (3.5%) that should be used for time discounting calculations within an English offsetting scheme.

67. **Figure 7** shows the multipliers that derive for a number of time periods using a discount rate of 3.5 %

Figure 7: Multipliers for different time periods using a 3.5% discount rate

Years to target condition	Multiplier
5	1.2
10	1.4
15	1.7
20	2.0
25	2.4
30	2.8
32	3

68. The following are the parameters within which the time discounting should operate for the biodiversity offsetting pilot.

69. The number of years that time discounting should take into consideration is from the point of impact to the estimated time that it will take for the habitat to reach the pre-agreed target quality (i.e. the point at which the agreed number of units is delivered). For simplicity and to allow upfront estimates of the offsetting provision this will require some guidelines. TEEB 2009 provides a good starting point, and **Appendix 2** has a table of estimated timescales from that publication. The actual figure will need to be calculated on a case by case basis for each offset management plan, taking into account the habitat type, and the amount of restoration or expansion being undertaken.

70. The calculations around the time discount multiplier should cover the whole period concerned. The calculations should assume that there is a quality jump from the baseline condition to the target condition once the relevant number of years has elapsed. The calculations therefore do not need to take into account increasing quality in the habitat, and do not need to be re-done annually.

71. Offsets should last at least as long as the impact of the development, and ideally in perpetuity. However, to be practical, there needs to be a limit on application of the discount rate used for time preference. We therefore propose that the maximum multiplier used to take account of temporal risk is x3.

72. We think that offset providers participating in the pilot should apply a temporal multiplier to their projects when calculating how many units of biodiversity they are able to offer.

Hedgerows

73. Hedgerows are a feature almost unique to the UK and there is no experience of dealing with them in offset schemes elsewhere that we can draw on. Hedgerows' contribution to biodiversity in the landscape is far greater per unit of area than even the most biodiversity rich habitats because of their role in provision of nest sites, corridors, feeding sites, shelter belts etc. They cannot simply be treated as other habitats and accounted for on a hectarage basis. It is therefore necessary to come up with a mechanism to account for hedgerows in our approach to offsetting that both recognises their unique contribution to biodiversity whilst at the same time meeting our guiding principle of simplicity.

74. Although this description is written to describe how we deal with hedgerows the conclusions and approach could equally apply, in theory, to other field boundary features such as hedge banks and rows of trees.

75. There is little if any science to draw on that compares the value of a hedgerow to other habitats. Even if such evidence did exist, it is likely that the exact value would be so dependent on a wide range of factors as to make its use as a generalisation difficult. Consequently it is recommended that hedgerows are treated as a separate case out with the main metric system.

A Guide to the Management and Restoration of Coastal Vegetated Shingle



By Dr Pat Doody and Dr Roland Randall

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Contract No. MAR 05-03-002 English Nature

A Guide to the Management and Restoration of Coastal Vegetated Shingle

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Frontispiece:
Dungeness - shingle extraction on the downdrift (eastern side) of the ness. The material is taken by lorries to nourish the beach to the west. Longshore drift moves it eastwards again providing protection for the nuclear power stations present on this tip of England - a never ending cycle of coastal protection?

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03 Pagham Harbour;
04 Rye Harbour;
05 Dungeness;
Annex 05a 'Living with the Sea' Executive summary Dungeness - Pett Levels CHaMP;
06 Orfordness;
Annex 06a 'Living with the Sea' Executive summary Suffolk Coast & Estuaries CHaMP;
07 Cley-Salthouse;
Annex 07a 'Living with the Sea' Executive summary North Norfolk Coast CHaMP
08 Spey Bay;
09 Culbin.

Site locations are shown on the Map opposite.



Chapter 01 Introduction and aims

Definition and description

Shingle is the term applied to sediments larger in diameter than sand but smaller than boulders. A predominant particle size of over 2mm separates shingle from sand (King 1972). At much over 200mm diameter they may be considered to be boulders and in ecological terms approximate to cliff habitat, i.e. their surface is more significant than the spaces between the individual pebbles. Around Europe such coastal habitats are variously known as *galets* (France), *cascalho* (Portugal), *playa de cantos* (Spain), *stenstrand* (Denmark), *steinfelder* (Germany), *klapper stens felt* (Sweden).

Four environmental factors are responsible for the growth of a shingle beach. There must be an available supply of sedimentary material at the same time as conditions of waves, winds and tidal currents are favourable for its movement. Since this coincidence is unpredictable, conditions without movement occurring may exist for considerable periods, interspersed by times of marked activity resulting in stable and mobile shingle habitats varying both in time and space (Randall 1988).

Geomorphologists and ecologists have recognised five categories of shingle structures (Chapman 1976) which vary in their oceanicity and therefore in their ecology. The simplest and commonest type is the **fringing beach** forming a strip in contact with the land along the top of the beach. **Shingle spits** form where there is an abrupt change in the direction of the coast. They often contain recurved hooks and a recurved distal end, a result of deflection of waves by refraction. This recurvature may also be caused by waves approaching from two directions. **Bars** or **barriers** are effectively spits, which have formed across estuary mouths or indentations in the coast. Ecologically bars differ from spits in having a less maritime environment to their lee (Kidson 1963). These three structures are basically foreshores, regularly washed by spray and storm waves and therefore possessing ephemeral vegetation which will vary in extent and frequency of establishment.

Cuspate forelands and **offshore barrier islands**, on the other hand, are larger structures and have a more terrestrial nature. The former develop when shingle is available in large quantities and piles up in front of fringing beaches or spits and is then driven landwards by storm waves to form apposition beaches. If this process is repeated, a series of roughly parallel ridges may develop and an extensive area of stable shingle results. Where wave approach can be from two directions only, such apposition beaches will form into cuspate forelands. Large masses of shingle may also form offshore barrier islands under conditions of shallow water and low energy environments such as in the North Sea and the Baltic. In practise it is often impossible to treat these features as separate entities as they are frequently intricately linked with dune or salt-marsh habitats.

The type of vegetation assemblages that may recur on shingle beaches and structures is strongly influenced by stability. As stability increases the number of species able to colonise the shingle surface increases, helping to create a wide range of community types, which are at their most complex on the most stable sites. The relationship between the type of shingle beach or structures and the plant communities is indicated below. The first three categories all tend to be associated with fringing beaches, spits, bars and barriers; categories 4 and 5 develop on shingle structures:

1. unstable beaches - lacking vegetation. Common in high energy, west coast situations;
2. beaches stable between spring and autumn - summer annuals (e.g. *Galium aparine* and *Atriplex* spp.);
3. beaches stable over 3-4 year periods - short-lived perennials (e.g. *Sedum acre, Desmazeria marina*);
4. beaches stable over 5-20 year periods - long-lived perennials (e.g. *Crambe maritima, Suaeda vera, Silene vulgaris* ssp *maritima*;
5. beaches stable over very long periods with heath or grass-heath vegetation (e.g. *Arrhenatherum elatius, Festuca rubra, Rubus fruticosus, Cytisus scoparius, Prunus spinosa, Calluna vulgaris.*

Besides stability (or its converse mobility) beach composition is the other principal factor affecting vegetation. There are four major types of shingle beaches, depending upon the nature of the fine material present as a matrix within the shingle. These are:

1. shingle without matrix - relatively less common, is limited to a vegetation of encrusting lichens and the most tolerant angiosperms such as *Lathyrus japonicus*;
2. shingle with a sand matrix - ecologically having the hydrological conditions of foredunes but the stability of rear dunes, contains species such as *Lotus corniculatus, Honckenya peploides*;
3. shingle with a silt-clay matrix - ecologically related to salt-marshes but with freer drainage. The vegetation is similar to that of saltmarsh levées with *Halimione portulacoides* or *Glaux maritima*;
4. shingle with an organic matrix. Rotting seaweed is nutrient rich and will support large populations of species that arrive as seed within tidal debris, particularly *Beta maritima* or *Atriplex* spp. where drainage from inland is poor, humic soils may develop and reed or iris swamp may be present.

Shingle acts as a habitat for both fauna and flora. The fauna associated with shingle structures largely consists of invertebrates (Philp and McLean 1985) and avifauna. The importance of shingle as a habitat for breeding terns has led to the conservation of several sites (e.g. Blakeney, North Norfolk and South Walney, northwest England). Few vertebrates other than domestic herbivores are found on shingle sites except hares and rabbits (White 1967), though in places such as Rye, foxes are important predators of ground nesting birds.

Location

Shorelines dominated by clasts of the size considered to qualify as shingle are important mainly in high latitudes and in those parts of the temperate world that were affected by Pleistocene glaciation. The combination of sediment production and supply and the post-glacial storm wave climate have helped to create gravel beach concentrations in the areas shown in Figure 01-01 (Orford et al. 2001). The figure is derived from this paper and Pye (2001).

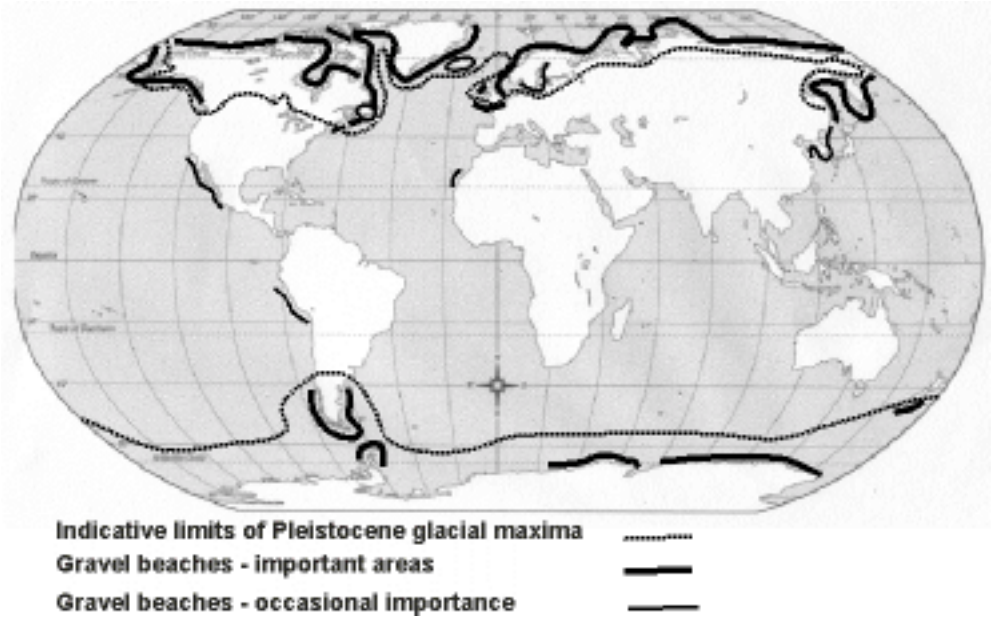


Figure 01-01 Approximate world distribution of gravel (shingle) dominated coastlines in relation to the limits of glacial activity

Europe

Within Europe, the occurrence of shingle structures has been documented for Britain (Randall 1989, Sneddon and Randall 1993). Géhu (1960 a, b, 1991) has described much of the shingle of western France; Bocher (1969) examined the vegetation of Danish shingle and Eklund (1931) gave an account of similar vegetation in southwest Finland. Nordhanger (1940) examined the vegetation of coasts of Norway where an organic matrix was present. Figure 01-02 provides an indication of their approximate location. A full literature review for the shingle habitat can be found in Sneddon & Randall (1989).

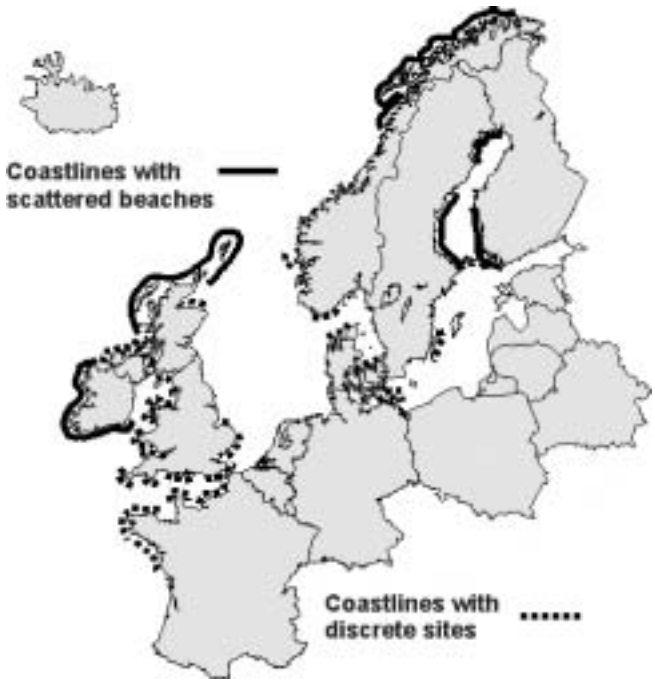


Figure 01-02 Approximate location of shingle beaches and structures in northwest Europe (various sources). Shingle also occurs in scattered localities around the Iberian Peninsula and in the Mediterranean

Systematic information is, as yet, lacking for much of Europe outside Great Britain (see below), but it appears that the most significant sites are on the Brest peninsula of France and the Rügen coast of Germany. Further detail is required. Coasts which are, or have been subject to glaciation or periglaciation often have plentiful supplies of rock fragments for the formation of shingle beaches; elsewhere other sediments predominate unless a shingle source (such as flint from chalk) exists locally.

Great Britain

Great Britain supports a significant proportion of European shingle shores structures, which are found extensively around the coast (Figure 01-03). The largest areas are in the northwest, south and southeast. The shingle in the northwest tends to be associated with beaches fringing sea lochs and in the south east major structures such as Dungeness in Kent. More detailed information on the location of shingle sites in Great Britain is presented in Chapter 04.

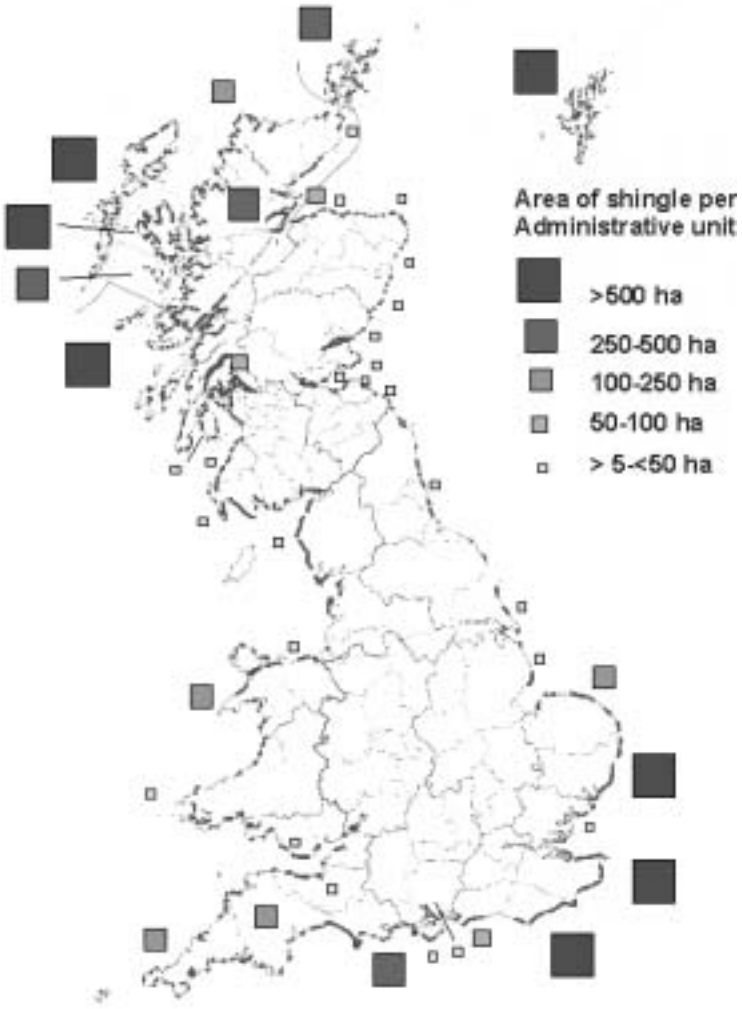


Figure 01-03 Distribution of shingle beaches and structures in Great Britain

Within the British shingle structures a series of distributional relationships (Table 01-01) can be seen among the major vascular plants, which are likely to be replicated and added to for Europe as a whole. If, however, the vegetation communities are analysed quantitatively, four major regional sectors are identified (northeast, southeast, south and west) with a less significant sub-set in the northwest.

Table 01-01 Distributional relationships of the major vascular plants of British shingle

Primarily southeastern	Southern	Western	Northeastern	Inland species with shingle ecotypes
Corynephorus canescens *Parapholis incurva* *Limonium bellidifolium* *Lathyrus japonicus* *Suaeda vera* *Frankenia laevis* *Lactuca saligna*	*Parapholis strigosa* *Limonium binervosum* *Rumex rupestris* *Trifolium scabrum* *Beta vulgaris maritima* *Tamarix gallica* *Glaucium flavum* *Eryngium maritimum* *Atriplex portulacoides* *Crambe maritima*	*Polygonum oxyspermum* *Sedum anglicum* *Coincya monensis* *Crithmum maritimum* *Polygonum maritimum* *Geranium purpureum* *Raphanus maritimus* *Lavatera arborea*	*Mertensia maritima* *Cochlearia scotica*	*Rumex crispus* *Solanum dulcamara* *Geranium robertianum* *Galium aparine* *Arrhenatherum elatius* *Cytisus scoparius*

Land Use, threats and conservation

In Britain, at least, most significant shingle structures are incorporated into Sites of Special Scientific Interest (SSSI), largely to protect the birds that frequent them and for their flora. Only about 10% of sites have no conservation designation. Approximately 200 Sites of Special Scientific Interest (Chapter 04) in Britain contain some shingle although in many cases this takes the form of a narrow fringing beach. Doody (1989) identified 22 as containing stable or semi-stable vegetation over significant areas. Three National Parks contain coastal shingle (Exmoor, Pembrokeshire and Snowdon). Several sites form part of National Nature Reserves, and several more are managed by local authorities as local nature reserves.

40% of the coastline of England and Wales is Heritage Coast and this assists in the protection of the north Norfolk shingle sites, the Suffolk coast and Dungeness. The north Devon coast and many of the Welsh sites are also proposed as Heritage Coast. In addition many shingle structures are part of Areas of Outstanding Natural Beauty (AONB) and the Royal Society for the Protection of Birds (RSPB) and the National Trust (NT) also own areas of shingle structures. The north Norfolk Coast and Bridgewater Bay are Biosphere reserves.

Despite these protection measures many sites are still at risk from development. Even within SSSI's the major threats to vegetated shingle structures are gravel extraction, coastal defence works, building construction, military use, agriculture, forestry and recreational pressure (Chapter 05). Ownership by the country Nature Conservation Organisations within the Joint Nature Conservation Committee, RSPB, National Trust and the County Conservation Trusts provides the most effective protection (Gubbay 1986).

Doody (1989) ranked 22 vegetation shingle structures in Britain subjectively according to size and degree of successional development (Table 01-02).

Table 01-02 Vegetated shingle structures in Great Britain within Sites of Special Scientific Interest (SSSI)

A:	**Sites of national importance**	**Approx. SSSI Area ha**	
1.	Dungeness, Kent	2714	
2.	Orfordness Havergate, Suffolk	1602	
3.	Blakeney Point, Norfolk	581	
4.	Chesil Beach, Dorset	1281	
5.	Culbin Shingle Bar, Moray/Nairn	12295	*
6.	Rhunahaorine Point, Argyll and Bute	327	
7.	Kingston Shingles, Moray	500	
B:	**Other sites of significance**		
1.	Bridgewater Bay, Somerset	2460	*
2.	Slapton Ley, Devon	219	
3.	Browndown, Hampshire	64	*
4.	Pagham Harbour, West Sussex	384	*
5.	Rye Harbour, East Sussex	721	
6.	Colne Point, Essex	3806	*
7.	Landguard Common, Suffolk	31	
8.	Scolt Head, Norfolk	738	*
9.	Walney and Foulney Islands, Cumbria	1909	*
10.	Cemlyn Bay, Gwynedd	45	
11.	Traeth Tanbwylch, Dyfed	36	
12.	East Aberthaw Coast, South Glamorgan	68	
13.	Whiteness Head Iverness/Nairn	409	
14.	Copinsay, Orkney	152	
15.	Ballantrae Shingle Beach, Kyle and Carrick	34	

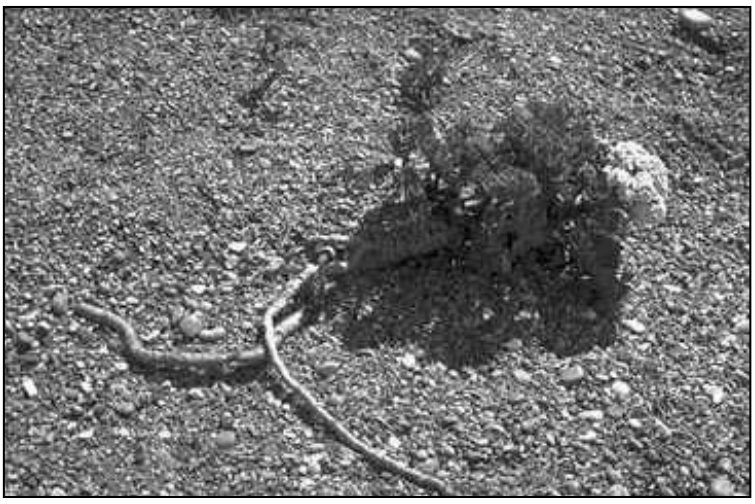
* Sites where the important shingle areas represent less than 10% of the total site area (After Doody, 1989)

If the most important sites are ranked objectively by floristic richness, size and disturbance factors (Sneddon 1992) it can be seen that the rank order changes quite significantly. In all three methods of ranking, however, Chesil Beach, Kingston Shingles, Rye Harbour, Slapton Beach and South Walney come into the top ten locations (Table 01-03).

Table 01-03 Shingle sites, the top 20, ranked by floristic richness (one most diverse), size (one largest) and disturbance (one least disturbed)

Rank	Floristic ranking	Floristic + size ranking	Floristic, size and disturbance ranking
1	Scilly Isles	Dungeness	Chesil
2	Dungeness	Rye Harbour	South Walney
3	Kingston	Kingston	Scilly Isles
4	Chesil	Chesil	Rye Harbour
5	Rye	Rhunahaorine	Blakeney
6	Slapton	Slapton	Kingston
7	Rhunahaorine	Snettisham	Whiteness Head
8	South Walney	Blakeney	Claymoddie
9	Snettisham	Browndown	Slapton
10	The Duver	South Walney	The Duver
11	Traeth TanybwIch	Orfordness	Orfordness
12	Browndown	Pagham Harbour	Snettisham
13	Blakeney	The Duver	Dungeness
14	Cemlyn	Whiteness Head	Colne Point
15	Crabhall	Scilly Isles	Pagham Harbour
16	Pagham Harbour	Traeth TanybwIch	Culbin
17	Orfordness	Claymoddie	Cemlyn
18	Whiteness Head	Colne Point	Rhunahaorine
19	Pwll du	Crabhall	Rascarrel Bay
20	Dougerie, West Arran	Shingle Street	Pwll du

Over Britain and indeed over Europe as a whole, climate is a major variable affecting shingle structures. Any form of climatic change would have implications for shingle communities. Associated hazards are related to accompanying sea level changes and increased frequency of storms. The loss of much of the raised shingle foreshore on West Arran in the storms of 1988 and the extinction of *Mertensia maritima* from the Isle of Man and Walney Island in 1967 (Randall 1986) are examples of this hazard. Boorman (1991) reports a median estimate of 0.8m sea-level rise over the next 100 years. Areas of British shingle which lie below 5m OD include all sites between the Suffolk coast and Browndown, Hampshire, along with the north Welsh coast, Foulney, Walney and the inner Solway Firth. Even if shingle is not lost, increased frequency of saline intrusion will lead to loss of heath species on terrestrial shingle as happened at Browndown and The Duver (Isle of Wight) in 1989.



A combination of factors have made rare or declining species quite characteristic of shingle structures with *Crambe maritima* (Scott & Randall 1976) Figure 01-04, *Lathyrus japonicus* (Randall 1977) Figure 01-05 and *Mertensia maritima* (Farrell & Randall 1992, Randall 1988) Figure 01-06 having been examined in detail.

Figure 01-04 *Crambe maritima*. Note the long tap root which helps stabilise the plant in the face of disturbance by waves.



Figure 01-05 *Lathyrus japonicus*, a plant of shingle beaches in the south and east of England



Figure 01-06 *Mertensia maritima* a species of undisturbed northern shingle beaches

Project aims

The aim of this contract is to produce good practice guidelines for the management of coastal vegetated shingle. It focuses on flood management works and their effect on shingle; covering the following broad subject areas:

- Physical and ecological attributes of coastal vegetated shingle;
- Importance of shingle structures in flood management;
- Nature conservation legislation;
- Threats to coastal vegetated shingle from flood management and other works;
- Guidance on managing threats to coastal vegetated shingle;
- Restoration opportunities;
- Monitoring.

The coastal vegetated shingle guide has been produced in two parts:

1. As a stand alone document - **The report**, which is reproduced here;
2. and as an interactive **Electronic guide**, which will initially be available on CD-ROM via English Nature in mid 2003.

1. The report

The report provides an introduction to the habitat, its attributes and guidance on management and restoration under the following Chapter headings:

Chapter 02 Describes the physical and ecological attributes of coastal vegetated shingle;

Chapter 03 Emphasises the importance of shingle structures in flood management;

Chapter 04 Provides information on shingle sites in Great Britain, including maps showing their location;

Chapter 05 Identifies the pressures on, and threats to coastal vegetated shingle;

Chapter 06 Provides habitat specific guidance on management;

Chapter 07 Highlights opportunities for habitat management and enhancement;

Chapter 08 Outlines monitoring requirements;

Chapter 09 Gives a summary of conclusions identifying any gaps in knowledge and provides recommendations for further work;

Chapter 10 Identifies sources of funding and useful contacts;

Chapter 11 Glossary and References;

Appendix 01 provides a summary of the legislation that may apply to managing and restoring coastal vegetated shingle

Appendix 02 provides a summary of the coastal vegetated shingle interest defined for the candidate Special Areas of Conservation in Great Britain

Appendix 03 give a list of invertebrates and their habitat preferences

Annexes 01-09 contain the site casework reports:

01 Porlock;
02 Slapton;
03 Pagham Harbour;
04 Rye Harbour;
05 Dungeness;
06 Orfordness;
07 Cley-Salthouse;
08 Spey Bay (Kingston shingles);
09 Culbin Shingle Bar.

2. An electronic guide

A contract has been let by English Nature to provide an interactive electronic guide to shingle habitat management and restoration. This will be linked to the Guide “Coastal Habitat Restoration, towards good practice” being prepared as part of the Living with the Seas LIFE Nature project. This will be available in July 2003.

Copies of the Executive Summaries of the CHaMPs within which sites 05, 06 and 07 are situated are included on the cd-rom.

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Chapter 02 Coastal vegetated shingle, geomorphology, natural processes and ecology

Chapter 02 provides an outline of the geology, geomorphology and natural processes of coastal vegetated shingle. This is followed by a review of the main features of the ecology of vegetated shingle, its vegetation, invertebrate and breeding bird interests. [N.B. The locations of sites in Great Britain mentioned in the text are shown in Figures 04-01a, b & c.]

Geomorphology

Introduction

The term 'shingle' has been used for at least 400 years in Britain and some Commonwealth countries, to describe sediments composed of mainly rounded pebbles, larger in diameter than sand (>2 mm) but smaller than boulders (<200 mm). Elsewhere terms such as gravel, stone, levées de galets, playas de cantos, schotterwälle and steinstrand are used. A generalised world distribution of shingle coasts is given in Figure 01.01 (Page 2). In many locations shingle is mixed with sand, silt, clay or organic debris, resulting in a 'mixed' sediment beach but all shingle and boulder beaches can be regarded as different types of 'coarse clastic' beach (Carter & Orford 1993).

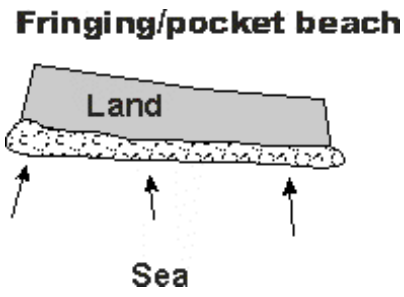
In general shingle coasts have received less scientific attention than sandy and muddy shorelines. In part, this reflects the fact that, at a world scale, they are much less common. However, in recent decades there has been an increasing awareness of the geomorphologic, ecological and engineering significance of shingle coasts in the contexts of sea-level change, flood defence and habitat conservation. Such coasts are now recognised as an internationally important, but disappearing resource (Packham et al. 2001).

Landform types and coastal processes

Shingle coasts form in wave dominated locations where suitably sized material is available. At a global scale they dominate high latitudes and those areas of temperate shores which were affected by Pleistocene glaciation (Figure 01-01). They are locally important in some other temperate and low latitude areas where high relief landscapes of suitable geology occur near the coast, near the estuaries of high-energy rivers, or where coral is present. Elsewhere they are of limited importance.

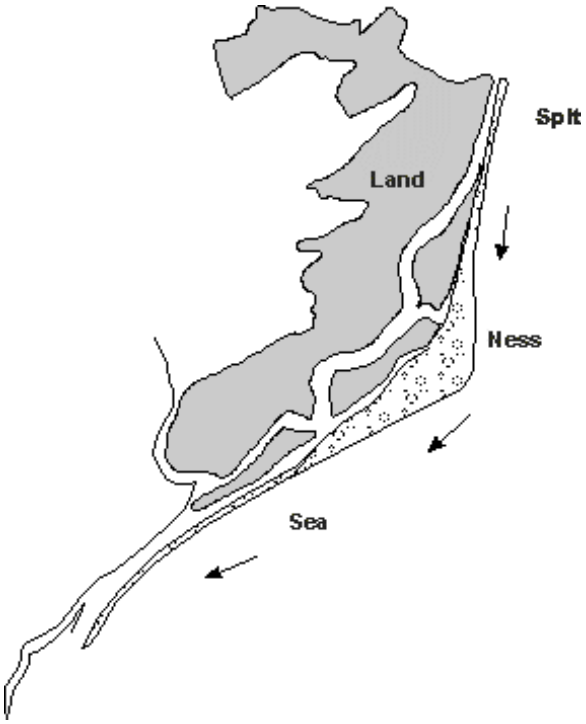
At a regional scale, lithographic composition determines shingle availability and durability. Hard materials such as flint, chert, granite, quartzite, and some metamorphic materials survive much longer at this clast size than sandstones, limestones or shells. Around Great Britain of the nearly 19,000 km of shoreline approaximately 30% has an important shingle component, with almost 3,500 km of these coasts being pure shingle (Sneddon & Randall 1993/1994). Many of the shingle-barrier systems occurring on present-day coastlines were initiated during the Holocene marine transgression and are currently sustaining considerable morphological change as a result of increasing sea-levels causing landward and longshore re-working of a finite sediment volume.

Shingle coasts can comprise several different landform types, which vary according to their history, mobility and oceanicity and therefore offer different habitats to vegetation, wildlife and man (Pye 2001, Sneddon & Randall 1993/1994).



Fringing or **pocket beaches** are narrow strips of shingle coast in contact with the land along the top of the beach. These are usually subject to regular marine inundation. They frequently occur at the foot of sedimentary cliffs such as chalk in southern Britain, but may also occur in front of coastal dunes or salt-marsh cliffs.

Embayment beach-ridge plains, or apposition beaches, are comprised of a series of relict storm beach-ridges and an active front ridge system which together partly or totally infill a previous embayment. Such systems may be hundreds of metres or even kilometres wide and can be transitional to cuspate forelands or nesses see below.



Shingle spits are strips of shingle, which grow out from the coast where there is an abrupt change in the direction of the coastline. They commonly occur, therefore, along coasts, which have an irregular plan. Spits often display recurved hooks along their length and at their distal ends, where the shingle is, or has been, subject to wave action from two or more directions. Indeed, in many cases, it is possible to trace the development of a spit's growth via recurved hooks, seen as lateral projections from the lee of the spit, which locate the position of the past distal points, (Randall 1973 and Figure 02-01). Paired spits are found at the entrance of several harbours on the south coast of England, including Pagham and Langstone. These may have originated as bars or tombolos, which have breached, but in other cases, independent growth of two spits may be due to bi-directional longshore drift.

On eroding coasts, shingle spits are transgressive and frequently overlay back-barrier marsh or lagoon deposits as at Shingle Street, Suffolk, and in some instances may be dissected to form barrier islands (Scolt Head, Norfolk also provides a good graphical example of this process). Transgressive ridges, often composed mainly of shell-shingle, are well developed on the saltmarsh coast of Essex. Similar features are also found in the Gulf of Mexico and in Auckland Bay, New Zealand. In Essex and the Gulf of Mexico they are known as 'cheniers' or chenier ridges respectively.

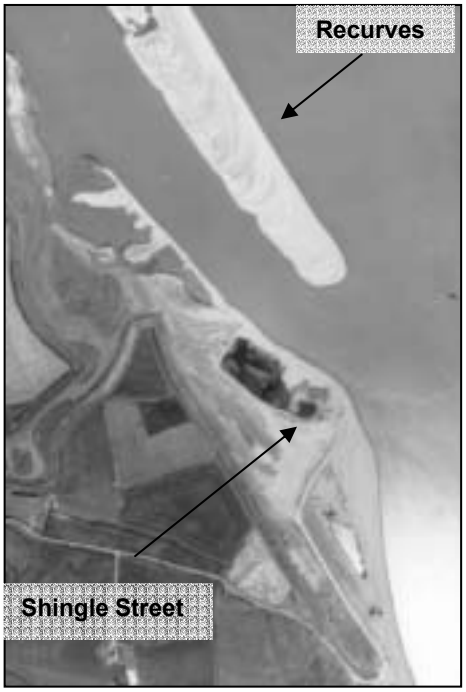
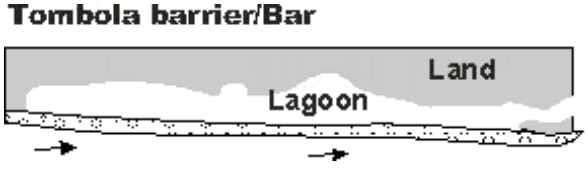


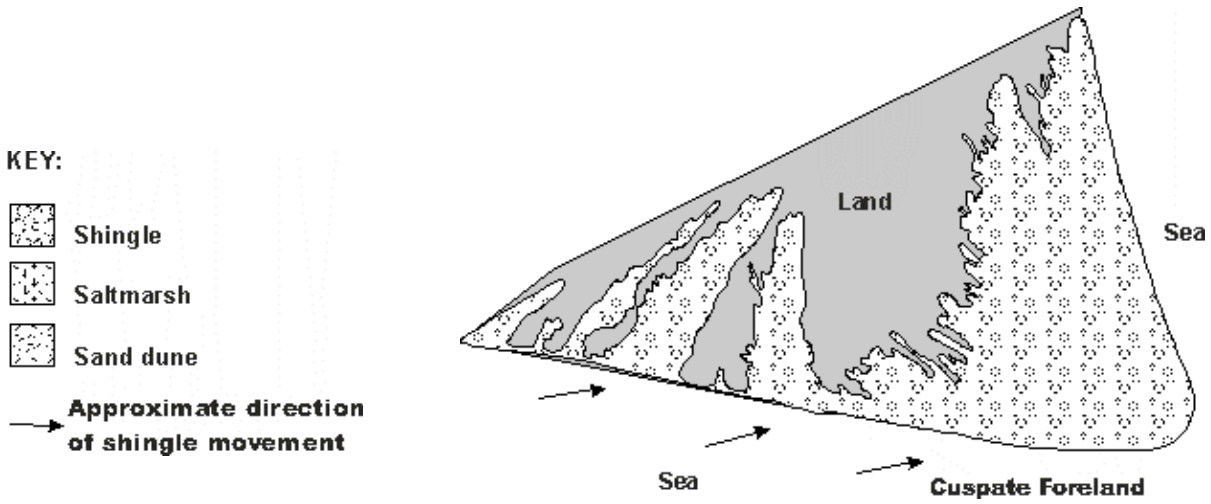
Figure 02-01 The shingle coast of Suffolk, UK. On the distal point of Orfordness and on the mainland opposite there are recurved hooks enclosing lagoons. At the mouth of the estuary of the River Ore longshore drift of shingle sediments can be observed. (Photograph RC8 D 121 - courtesy of University of Cambridge Committee for Aerial Photography.)

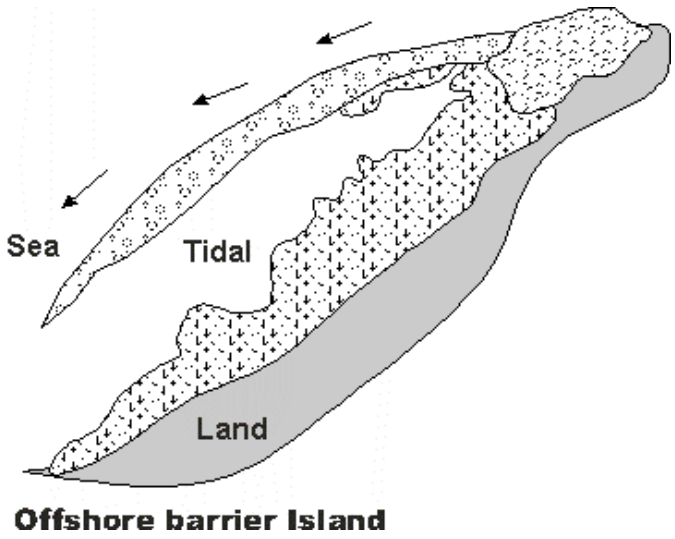
Tombolo barriers, or **bars**, are geomorphologically similar to spits, representing the extreme case where a spit has grown across an estuary or coastal indentation. This results in the formation of a lagoon behind the bar, which clearly affects the hydrology, and ecology of the leeward slope. Chesil Beach in Dorset is a prime British example. Rivers, which provide a source of shingle-sized sediment, may have prograded strandplains or deltas of shingle at their mouth. In Scotland the Kingston Shingles are found at the mouth of the Spey (Sneddon & Randall 1993/1994) and South Island, New Zealand has particularly good examples, such as at the mouth of the Waitaki River.



At points of littoral drift convergence the formation of a second set of apposition ridges deposited at a different angle, will lead to the formation of a ness or **cuspate foreland**, a triangular mass of shingle such as Dungeness, Kent, Rhunahaorine Point, Argyll or Cape Canaveral in Florida. The Island of

Rügen in Baltic Germany is effectively a cuspate foreland cut off from the mainland. Such features often support a terrestrial geomorphic system inland of the coastal ridges.





The final type of shingle formation is the **offshore barrier island**, formed where a large mass of shingle has been deposited offshore and which may act as the 'skeleton' for a coastal sand-dune system. Culbin Bar, Morayshire and Scolt Head Island, Norfolk, are prime examples.

Most shingle coasts have a steep upper beach slope and a relatively steep overall nearshore profile. Partial wave energy reflection results in the formation of edge-waves and rhythmic longshore features such as beach cusps. Shingle features are frequently of considerable ecological importance in terms of habitat diversity and play a vital geomorphologic role in determining the stability of adjoining 'soft' sediments of mudflats and salt marshes. Unless the shingle coastal features are mobile, a partial vegetation cover is the norm. The middle and lower beach are usually kept bare by wave action, but upper beaches are vegetated. The rate and extent of plant colonisation is dependent upon the degree of disturbance and shingle mobility, the presence or otherwise of a fine sediment matrix within the spaces between larger sediments and the hydrological regime of the shingle.

Sediment patterns

All shingle coasts contain a mixture of different sized sediments. Some are well sorted and consist entirely of pebbles, while others are poorly sorted and may also contain sand and/or boulders. Because there is frequently considerable temporal and spatial variation in shingle and mixed shingle/sand beaches, accurate determination of average textural qualities is difficult.

Most coarse sediment coasts become coarser up-beach, because backwash and gravity cannot move larger clasts. Hence many locations have shingle only on the upper beach. Discoid pebbles are sorted preferentially on the upper parts of the beach with spheres and rods occurring nearer the sea. Sediment grading along-shore also occurs due to selective transport of finer sediments in the down-drift direction as at Chesil Beach. However, other sites show much more complex patterns as a result of bi-directional currents of varying magnitudes.

Shingle coast micro-relief dynamics depend upon spring to neap tidal patterns and wind and wave conditions. The upper 50-80 cm of sediment is frequently remobilised forming berms and cusps that change from one tidal-cycle to another. More major changes occur seasonally as a result of spring to neap tidal fluctuations and especially at those times when storm-wave energy is higher.

The internal sedimentary architecture of shingle landforms reflects the process regime and net evolutionary trends of the structure (Randall 1973). The external structure of ridges vary depending on whether they are vertically accreting but laterally stable, laterally migrating or developing on a seaward prograding plain. The depressions between ridge crests may be partly filled by washover and storm-tossed deposits, so that there is often a marked difference in average particle size and shape between ridge fulls (crests) and lows (Randall & Fuller 2001). Sediment grading may also occur as a result of longshore drift with selective transport of finer sediments downdrift. However, on many coasts sediment grading has been found to be complex in relation to seasonal variation in the longshore current regime (Pye 2001).

Sea-level change

Sea-level rise has the tendency to move shingle landforms inland (Carter & Orford 1993; Forbes et al. 1995), but if sea-level rise is particularly rapid, shingle structures may be drowned in situ by overstepping. Normally, however, under moderate storm-wave activity, shingle is pushed to the top of the front-beach ridge ('overtopping'); while in major storms the ridge is 'overwashed' (or even breached), creating shingle fans or aprons in the backbarrier area (e.g. Bradbury & Powell 1992). As this pattern is repeated, so the ridge migrates landward by rollover. Many of the major shingle formations present today formed in this way during the Holocene marine transgression, initiating at a time of lower sea-stand and reaching their present location by around 4000 BP. Most current shingle features are relict or dependent upon erosional sediments rather than glacial debris. Hence, there is currently a shortage of sediment at the updrift end of many transport cells and increasing risk of overwashing and breaching (Orford et al 2001).

Hydrology and soils of coastal shingle

Hydrology

Burnham & Cook (in Packham et al. 2001) reported that substantial areas of coastal shingle support a fresh water table that overlies saline water near the coast. Dungeness, Pevensey Beach, Blakeney Point and Shingle Street are good examples but there is little doubt that the hydrology is similar elsewhere. The significance of shingle as an aquifer is the great rate at which water passes through the coarse sediment. At Denge Beach, Dungeness (Annex 05) this ranges from 300-1000 m2/day. Such easy transmission means that any 'doming' of the water table is of low amplitude and water levels fluctuate with the tide. Penetration by sea water during storm tides can be equally rapid. Water abstraction from the Denge aquifer has taken place since the early years of the twentieth century. Following a report in 1984 that highlighted saline incursion this aquifer was seen as a finite and fragile resource. The abstraction licence was reduced to 3300Ml/annum and a management regime incorporated. Penetration by sea water during storm tides can be equally rapid.

Soils

Most shingle substrates do not develop a true soil. However, on shingle that has been stable for a very long period of time a 'ranker' or 'ranker-like' alluvial soil may develop. The fine component is primarily plant-derived and earthworms are normally absent. The development of vegetation seems to be associated more with the presence of fine shingle rather than elevation or exposure. The presence of fine particles is also important. Using the studies at Orfordness and Dungeness, Fuller (1987) suggests that the distribution of vegetated shingle is determined by a number of factors:

1. coarse shingle - seeds fall too deeply for germination and growth to the surface to occur. (*Note gardeners often use shingle to prevent weed growth*);
2. poor water retention of the shingle substratum inhibits germination and seedling establishment;

3. even if plants survive the initial stages of growth they may succumb to drought conditions and die, with the exception of a few highly specialised plants with long tap roots such as *Crambe maritima*;

Where plants do become established they help promote the build up of material through the decay of roots and above ground material. Mites and collembolans can help break down these plant remains to dark molder-like humus. At the same time the plant's ability to trap wind-blown debris increases litter accumulation. The soils so developed tend to be naturally highly acid, but can be nearer neutral if the shingle contains a large amount of shelly material. This accumulation of humus enhances moisture retention and with it the nutrient status of the soils and hence helps the establishment of more permanent plant cover. It is perhaps no accident that the more mature vegetation of the Culbin Shingle Bar (Annex 09) which is located in an area of high rainfall supports low-growing scrub in a matrix including *Salix repens*, *Empetrum nigrum* and *Calluna vulgaris* (see Figure 07-04). At Dungeness the soil mollusc fauna is substantial where calcium levels are higher as a result of input of shelly material.

Vegetated shingle

Vegetated shingle is recognised as an internationally important habitat. It is scarce in Europe with the UK supporting a high proportion of the European resource. Elsewhere, coastal shingle has few occurrences outside Japan and New Zealand. Coastal vegetated shingle is listed as a priority habitat in the UK Biodiversity Action Plan (BAP) and supports 9 BAP priority species. It is characterised by specialised plants that have adapted to survive in harsh coastal conditions where lack of fresh water and nutrients are compounded by fierce winds and impact by waves. Shingle habitats are also particularly important for invertebrates and for some breeding and roosting birds. Much has already been lost to housing developments, agriculture and coastal defence while the remaining area faces a number of threats including trampling and unnatural enrichment of the shingle substrate. One of the main long-term threats to vegetated shingle is as a result of man's intervention in natural coastal processes, with coast protection work changing the accretion rate of shingle to coastal areas. Trapped between urban development on the landward side and rising sea levels on the seaward side, vegetated shingle is also threatened by 'coastal squeeze'.

Shingle banks form a natural coastal defence, which may require replenishment in order to maintain the bank crest height and width. With coastal protection techniques moving towards a more integrated approach allowing natural processes to work where possible and relying more on 'soft' defences rather than 'hard' structures such as sea walls, there is an opportunity to create new, stable areas of shingle. If planned with care, these could be used to create new areas of shingle vegetation. However, it does not always follow that new areas of shingle will restore the plant and animal assemblages of areas that have been lost.

Vegetated shingle communities

The frequency with which shingle beaches are disturbed by the action of the sea varies according to wave fetch and prevailing meteorological conditions; and the resultant vegetation is similarly altered (Randall 1977). In practise the majority of shingle foreshores are unvegetated or have extremely sparse vegetation cover. Scott (1963) recognised three foreshore stability classes which can be observed on a vegetation basis, dependent on the length of time over which the shingle is undisturbed by environmental factors:

- no vegetation – disturbance too frequent to support plant growth: as at the foot of sea-cliffs, distal points of spits, high-energy beaches etc.;
- summer annuals – beach stable over growing season only: mainly *Galium aparine* and *Atriplex* spp. on drift line;
- short-lived perennials – beach stable for +3 years: considerable strand and foreshore vegetation *e.g. Glaucium flavum, Rumex crispus, Beta maritima.*

These three stability habitat classes, which are similar to those identified above in Chapter 01 in relation to the geomorphological types of shingle beaches, may occur at different levels on the same

beach. For instance at Shingle Street, Suffolk, the exposed foreshore is unvegetated, the lagoon foreshores have drift lines supporting *Atriplex* spp., whereas the main ridge crest with more stable shingle supports growth of *Lathyrus japonicus, Beta maritima* and *Rumex crispus*. On some shingle foreshores as at Cley (Annex 07), Norfolk, mobility of the substrate results in accretion around plants of *Suaeda vera.* Various authors have discussed the role of this species in shingle stabilisation. Because of the instability of many shingle foreshores they usually have low volumes of animal life. However, numbers of amphipods and collembola may be high at a few sites, such as at Dungeness.

Although the geomorphic classification of shingle is determined by predominant particle size, the vegetation is primarily controlled by the proportion and size of the fine fraction material of the matrix under 2mm diameter. In fact, because this is the main source of nutrients, it is commonly recorded that, even on stable beaches, the absence of a fine matrix results in a marked reduction in vegetation. Early workers suggested the importance of the fine fraction and this has also been emphasised frequently since. The fine fraction is critical at germination and seedling stages (Chapter 06) since, without it, enough moisture may not be present for growth to be initiated or to continue. The matrix is usually composed of sand, silt or organic matter, each type having distinctive vegetation. Within the British Isles the matrix tends to vary regionally: silt and clay are dominant in the south and east, sand in the west and organic matter in the northwest. Randall (1977) showed the plant species most commonly found with different shingle matrices. These are listed in the following table.

Table 02-01 Species commonly found on shingle with different types of fine fraction matrix

None*	Sand	Organic	Silt/clay
Arrhenatherum elatius+	*Festuca rubra arenaria*	*Urtica dioica*	*Beta vulgaris ssp. maritima*
Mertensia maritima	*Sedum acre*	*Tripleurospermum maritimum*	*Glaux maritima*
Silene uniflora	*Elytrigia juncea*	*Galium aparine*	*Artemisia maritima*
Geranium purpureum	*Honckenya peploides*	*Potentilla anserina*	*Puccinellia martima*
Geranium robertianum+	*Corynephorus canescens*	*Rumex crispus var. trigranulatus*	*Parapholis strigosa*
Solanum dulcamara+	*Desmazeria marina*	*Iris pseudacorus*	*Limonium bellidifolium*
Limonium binervosum	*Calystegia soldanella*	*Armeria maritima*	*Suaeda martima*
Polygonum maritimum	*Polygonum oxyspermum*	*Atriplex* spp.	*Atriplex portulacoides*
Crithmum maritimum	*Eryngium maritimum*		
Sedum anglicum	*Hippophaë rhamnoides*		
Lathyrus japonicus	*Rosa pimpinellifolia*		
Vicia lutea	*Trifolium scabrum*		
Suadea vera	*Atriplex laciniata*		
Atriplex glabriuscula	*Cakile maritima*		
Crambe maritima	*Silene conica*		
Glaucium flavum			
Lactuca saligna			

Water availability is another basic factor in the ecology of shingle foreshores. This is likely to be extremely low because of the high porosity and low water-retention of the substrate. Inefficient capillarity in shingle usually rules out the water table as a moisture source for all but the deepest rooted of shingle foreshore plants. Thus, the principal source of supply must be pendulant water from precipitation. However, the speed of water movement is again related to fine fraction content and diameter. Early workers suggested that further moisture supplies are obtained within shingle by internal dew formation but it was demonstrated that it was no more important than in sand. Of greater significance is the 'mulching' effect of large shingle particles on the soil surface, which causes a reduction in the evaporation of any water present in the upper layers.

There are a few plant species that are more characteristic of shingle than of other environments, most of them being associated with the extreme mobility of the foreshore zone. One species apparently exclusive to maritime shingle in Britain is *Suaeda vera*, which is limited climatically to the southeast. The plant is unusual in shingle foreshore habitats in that it is woody and upstanding, reaching over 1m in height. It is also evergreen in the optimal parts of its range. Usually *S. vera* germinates in the driftline rapidly sending long roots deep into the shingle, so stabilising the plant. Overwhelming by

shingle in storm conditions forces the plant to a horizontal position from which it sends out new roots and new vertical shoots.

In contrast, *Mertensia maritima* is a northern element in British shingle foreshore habitats. Typically it grows as a low cushion on 'pebble-wrack-sand' beaches on the north and west coasts of Scotland, where grazing does not occur. Good examples are The Churchill Barrier, Orkney and around the Castle of May, Caithness. An interesting change in the distribution of this species over recent years has been the decline or disappearance of its more southerly sites, possibly a response to climatic change. The fact that this species is highly susceptible to grazing and trampling may have also played a part. These factors have certainly been significant in the changing distribution of *Lathyrus japonicus. Glaucium flavum* and *Crambe maritima* are other characteristic species of shingle foreshores.

Although there is considerable variation in the assemblages of species found in shingle foreshore habitats, lists compiled from widely separated sites show some floristic pattern. Fringing beaches and the seaward slopes of spits and bars show constancy of several species, in particular *Tripleurospermum maritimum, Silene maritima, Atriplex glabriuscula, Rumex crispus* ssp. *littoreus* and, in the north, *Galium aparine.* Other species that are common in this habitat include *Festuca rubra, Beta maritima* and *Honckenya peploides.* Less common, but still locally important are *Potentilla anserina, Sonchus arvensis, Rumex acetosa, Elytrigia* spp*., Sedum acre* and *Senecio* spp. It will be noticed from this list of species that the majority, whether annual or perennial, are open ground nitrophiles. Their communities fall within the two Annex 1 habitats of the EU 'Habitats' Directive - the annual vegetation of drift lines and perennial vegetation of stony banks. However, as with more terrestrial shingle vegetation Sneddon and Randall (1993) showed that the communities around Britain are more complex than this twofold division suggests.

Landward slopes of spits and bars contrast with seaward slopes in that they are usually less mobile and have different nutrient inputs. Bars and spits, which have a poverty of drift material, are not so rich in flowering plants and many of the nutrient-hungry species are absent. On the north and west coasts where water tables may be higher, *Iris pseudacorus* and *Filipendula ulmaria* are common, whereas in drier conditions in the south and east *Geranium robertianum* and *Solanum dulcamara* occur in their coastal ecotypic form. On wider fringing beaches and some spits and bars on lower energy coasts *Tamarix gallica* and *Lupinus arboreus* have been introduced into this habitat. These species act as good protection for exhausted migrant birds on passage. In other areas, especially where spits are adjacent to salt-marsh, the reduced mobility and much-increased organic matter content of the substrate gives rise to narrow bands of halophytes: *Sarcocornia perennis, Salicornia* spp., *Suaeda maritima, Puccinellia maritima* and *Atriplex portulacoides* being the most frequent. Details of the variation around England, Scotland and Wales can be seen in Sneddon and Randall (1993/1994).

Much less common around Britain are the more terrestrial shingle formations of apposition beaches, cuspate forelands and offshore barrier islands. All these have typical foreshore habitats near the shoreline but further inland they are more stable. The largest apposition beach structures occur on the Isles of Arran and Jura; Dungeness, Kent has the most complete cuspate foreland and Culbin Bar and Scolt Head Island are the most studied barrier islands. Ecologically these features are of most interest because of the differences in the time-scales of their stability.

Because beach mobility is less of an ecological consideration on these larger-scale formations, Scott (1963) recognised two further vegetation classes relating to terrestrial shingle:

- long-lived perennial species – beach subject to occasional inundation, lichens present;
- heathland – beach entirely stable.

Vegetation succession

Understanding the factors determining the nature of vegetation succession is important when considering management and restoration options. This will be particularly significant when attempting to predict the outcome of a specific management operation. It is generally accepted that a greater expanse of shingle and a more stable formation result in a more complex ecological development over time. This has been studied by Ferry (in Packham et al. 2001) at Dungeness and Randall and Sneddon (in Packham et al. 2001) elsewhere in Britain. The quantity and composition of the matrix remains a vital factor throughout the successional sequence on stable shingle formations. It is of greatest importance at times of seed germination because, without it, enough moisture may not be present for growth to begin or to be sustained. The major difference between shingle foreshores and the larger, more terrestrial formations is that the latter do not have the advantage of external inputs of organic matter through tidal drift. Small quantities of nutritive material will be blown into these habitats by wind but most organic matter will be produced over long periods *in situ*, by the plants themselves. This hypothesis was first tested by Scott (1963) who later produced a tentative successional sequence based on Dungeness foreland (Scott 1965). Nine stages were recognised in an autogenic xerosere, ranging from bare shingle to climax woodland (Figure 02-02).

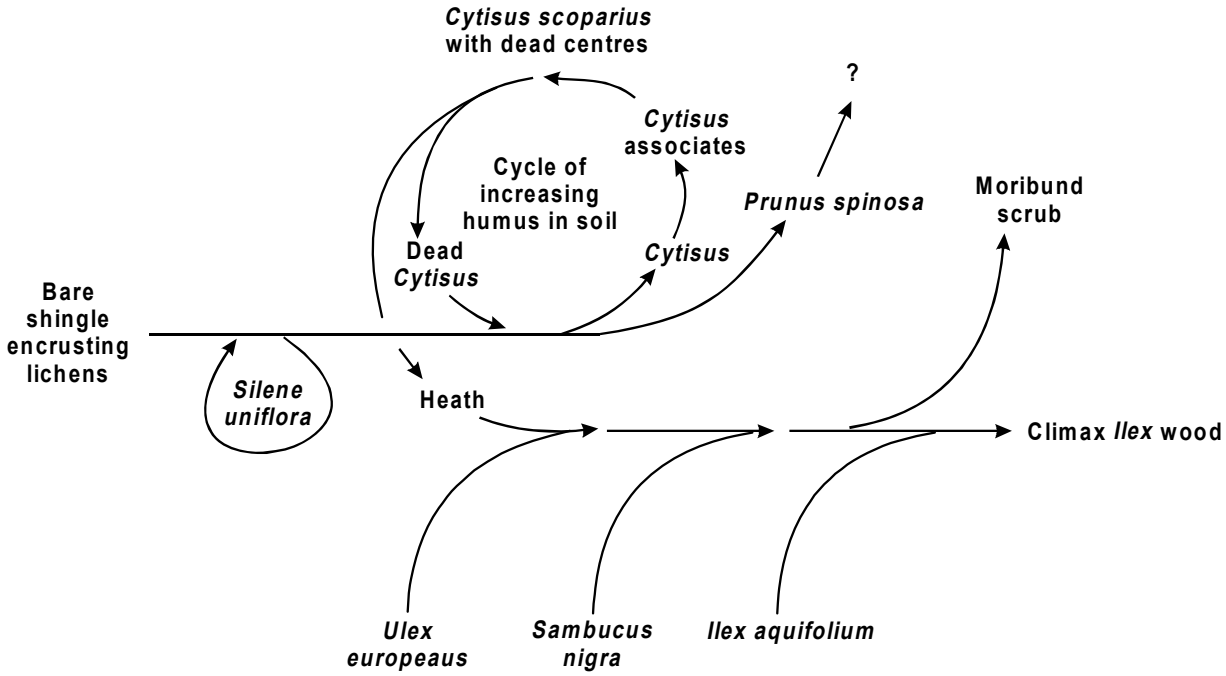


Figure 02-02 'Broom cycle' at Dungeness leading to a supposed holly wood climax (after Scott 1965)

Scott (1965) identifies a small-scale patterning of vegetation confined to the 'fulls' of shingle ridges, which support patches of dry, acid heath. This patchiness is recognised as supporting Whittaker's (1957) view of a sere as a general trend in vegetation change of 'loosely ordered complexity', conforming to Gleason's (1926) 'individualistic concept of the plant community'. Despite this, Scott notes a tendency for younger stages of succession to occur nearer the sea on more recently deposited shingle and older stages to landward, reflecting the known geomorphic sequence. Scott links the successional sequence to increasing humus within the shingle matrix, resulting from the dominant role of *Cytisus scoparius*, leading eventually to *Ilex aquifolium* woodland regarded as the climax (Figure 02-02). [*Teucrium scorodonia* is also important as a producer of woody humus.]

Peterken & Hubbard (1972), also working at Dungeness, question the status of *Ilex* and propose modifications of Scott's model in which various heathland species may increase humus within the soil and several scrub species are involved in the climax cycle dependent upon factors such as longevity, shade tolerance, fertility or dispersal mechanism (Figure 02-03).

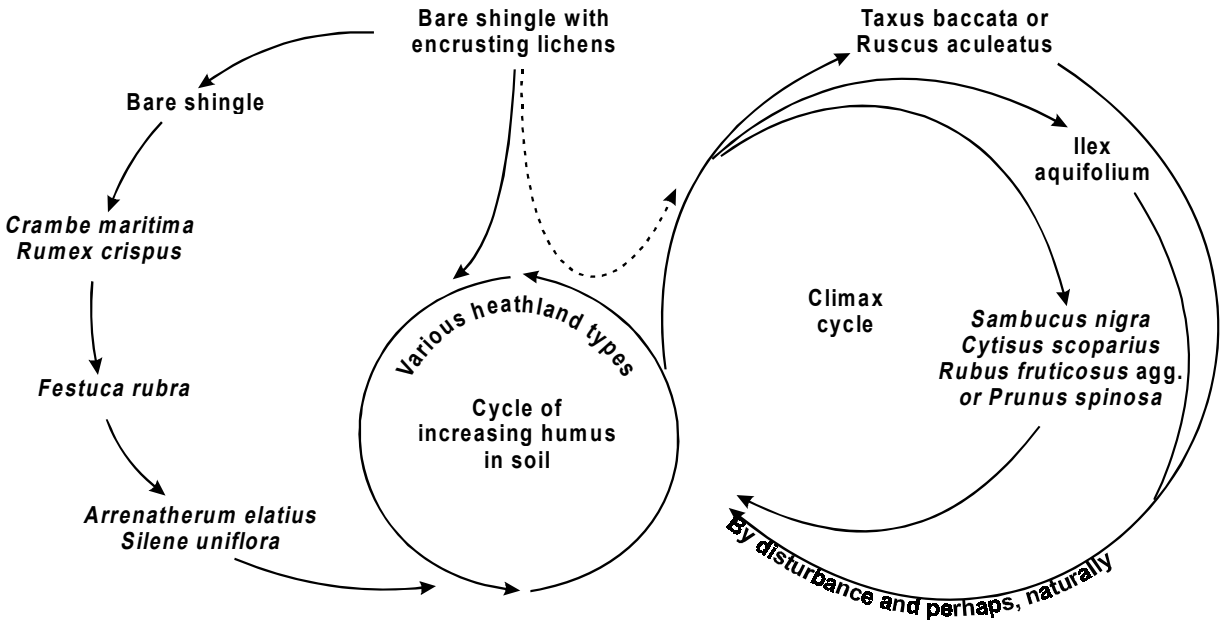


Figure 02-03 Cyclical and successional shingle vegetation at Dungeness (after Peterken & Hubbard 1972)

Ferry et al. (1989) refute these deterministic models and highlight the importance of stochastic events. Randall (1992), working in New Zealand also doubted the applicability of linear species models and suggests an anastomosing successional sequence with a development from herb to low, then taller, shrubs while the species differ considerably in each locality.

Connell & Slayter (1977) emphasise the *mechanisms* of succession, including 'facilitation', as being most significant, rather than the outmoded idea of climax. On shingle, pioneer vegetation has been shown to be particularly important in facilitating the provision of shelter for later species in the sequence. It also helps to stabilise the system and provides humus. The adoption of an 'anastomosing' sequence of succession, where the above ground plant material (of various species) breaks down to fill in the gaps between the pebbles, supports the development of a partially deterministic model. This model also allows for the incorporation of stochastic events. This type of model offers the best means of developing a useful predictive tool with wide applicability for management purposes. Full details are given in Randall & Sneddon (2001) and see also Figure 02-04 below.

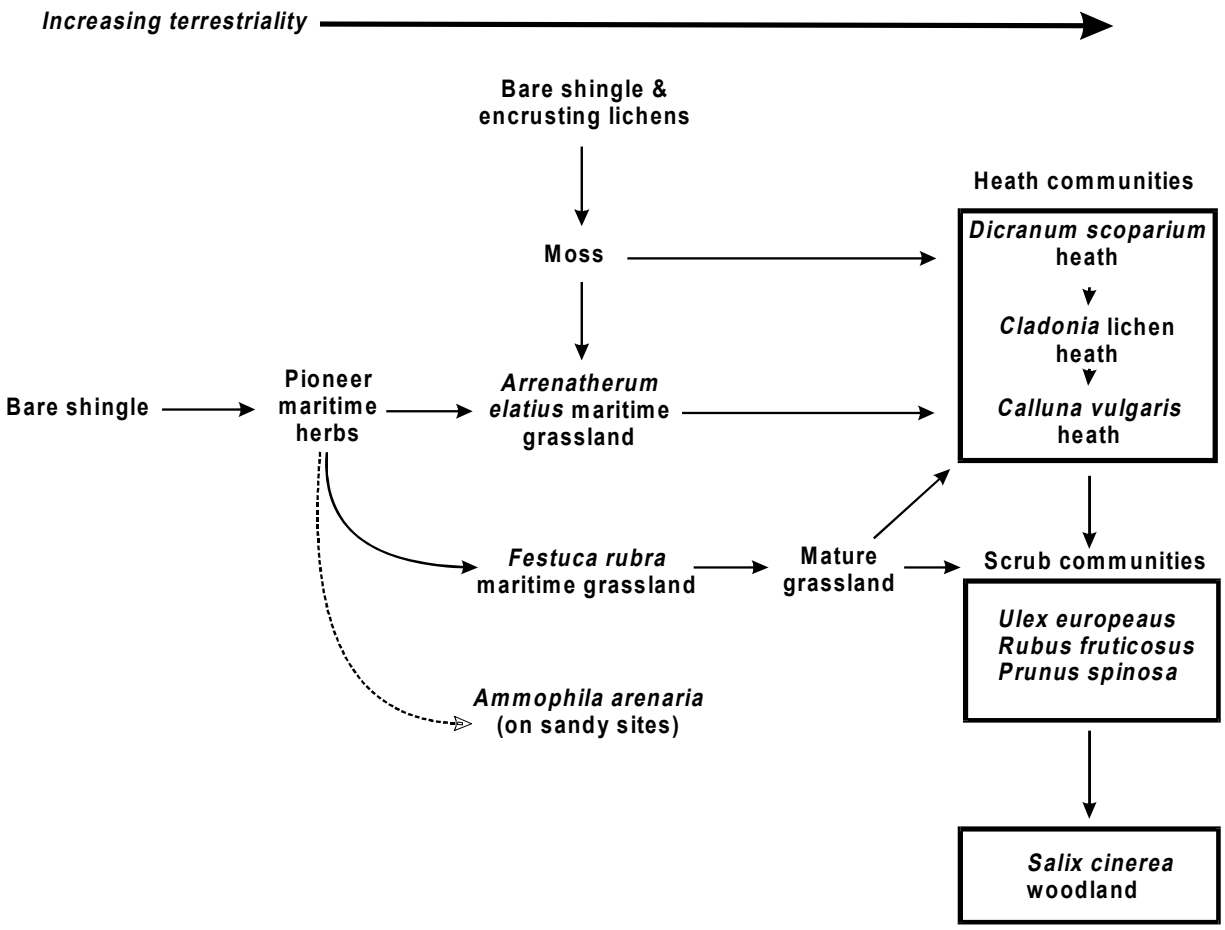


Figure 02-04 Proposed sequence of vegetation succession on shingle sites in Great Britain

Distinct communities also occur around artificial pits created by excavation and around lagoons that have developed naturally on shingle. The areas support a completely different flora dependent upon whether the water table is affected by fluxes of seawater or tidally related fluctuations in fresh-water level. Where seawater is able to seep through the shingle, lagoon waters are brackish and a limited number of saltmarsh species occur. *Artemesia maritima, Aster tripolium, Limonium spp. and Atriplex portulacoides* are the most frequent in these conditions. Where the water is less brackish there is usually a zonal sequence from open-water species through *Phragmites australis* to *Triglochin palustris* near the water's edge. Gimmingham (1964) also refers to 'shingle/salt-marsh communities' in western Scotland maintained by percolation of seawater through the storm-crest into shingle with a silt matrix. Here *Armeria maritima, Puccinellia maritima* and *Plantago maritima* form an intermittent, eroded turf over shingle.

A final community type is that of organic foreshores. These occur where, almost regardless of substrate, there is a very large algae deposit each winter, which does not fully decompose during the succeeding summer. Such conditions occur on high-energy coasts where the main constituent of the organic material is *Laminaria* spp. These areas mainly support communities of annual plants but some perennials can grow through further additions of organic debris. Over much of western Scotland *Atriplex glabriuscula* is the dominant species though *Potentilla anserina* is the major species in Mull. Such communities are dependent upon the chance input of seed and in different years many ruderal species may occur. The most frequent are: *Urtica urens, Poa* spp., *Polygonum* spp. and *Spergularia arvensis*. These strand communities are normally open, variable and ephemeral, without the more formal structure found elsewhere.

Invertebrate communities on shingle

Shardlow (in Packham et al. 2001) regards coastal shingle as a unique and fascinating habitat for invertebrates. He recognises two distinct habitats: saline-shingle beaches and terrestrial shingle. Some 390 species of importance to conservation have been recorded on shingle, including 15 Priority species, 18 species of Conservation Concern and 114 Red Data Book species. At least 11 shingle-specialist taxa occur in the UK, four of which are endemic. The current key factor threatening shingle invertebrates is thought to be changes in coastal sediment dynamics, related to management of the coastline and sea-level rise. In certain locations visitor pressure is also highly significant.

Barnes (in Packham et al. 2001) and McArthur (in Packham et al. 2001) look particularly at the ecology of shingle-enclosed lagoons, which are a priority habitat under Annex 1. Most lagoons are protected under legislation for other purposes but there is concern that our scanty knowledge of the inter-relationships between lagoonal organisms will make the long-term safeguarding of populations difficult. A list of invertebrates associated with coastal vegetated shingle is given in Appendix 03.

Birds of coastal shingle and lagoons

Birds associated with shingle can be divided into three groups. The first occurs on dry open shingle and is largely composed of about nine breeding species: oystercatcher, ringed plover, lapwing, grey partridge, and six passerines. Most of these occur more abundantly elsewhere. Stone curlew and Kentish plover formerly bred on coastal shingle but are no longer present. Wintering snow buntings use the east coast shingle beaches as a place to search for food.

Cadbury & Ausden (in Packham et al. 2001) regard shingle as more significant for two other groups: nesting colonial seabirds and waterfowl. Five species of gull, including the rare Mediterranean gull, four terns and cormorant breed on shingle. Many sites are particularly important for tern nesting and Orfordness has one of Britain's largest colonies of lesser black-backed and herring gulls. Among the waterfowl, avocet is restricted as a breeding species to coastal lagoons. Nationally high numbers of little grebe, gadwall and, at Dungeness, smew congregate on shingle water bodies in winter and Chesil Fleet is famous for its large flocks of brent geese and mute swans which graze on the beds of *Ruppia* and *Zostera.*

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Chapter 03 Flood defence and coastal protection - the role of shingle beaches and structures

Introduction

Shingle beaches are mobile structures developed in dynamic high-energy environments. As such they are themselves highly efficient dissipators of wave energy and may form important sea defences, a role that has long been recognised. For example, discussions over shingle extraction during the 1907 'Royal Commission on Coastal Erosion and the reclamation of tidal lands in the United Kingdom' prompted exchanges such as the following:

"Do you recommend that the removal of shingle, whether for manufacture of concrete, road-making, or ship ballast should be stopped?"

"I think that any beach that can be shown in any way to protect the coast should be left alone .... I think that ought to be enforced very strongly indeed. In many cases shingle is taken from comparatively narrow, small masses of beach, the decrease of which leads to very serious results, that is to say, the damage done is many times the worth of the shingle taken." [Vol. 1(2); page 92; minute 2268].

Despite this understanding, building sea defence structures (e.g. sea walls and groynes) has occurred on or behind many shingle shores. These can cause major change to the sediment regime, reducing or preventing available sediment reaching the beach. In its turn erosion due to the interruption of long-shore drift can result in the need for protection further along the beach. Reprofiling the beach also takes place destroying any incipient vegetation. This last point is particularly important for designated sites as most problems occur due to ongoing beach recycling operations carried out by "competent authorities". These often involve activities, which have had a significant impact on shingle vegetation. In some cases the integrity of the structure is affected and low-lying land protected by shingle beaches or structures may become vulnerable to flooding. To some extent the value of shingle to coastal defence is not recognised until it is too late, as for example the case of Hallsands (Annex 02) vividly illustrates.

Shingle as a natural defence

Sea defence (the protection of land from flooding) and coastal protection (protection of land from erosion) can both apply to shingle beaches and structures. Most of the shingle types identified in this report provide some form of flood defence. The shingle beach itself is a buffer to waves absorbing the energy of storms. Often the height of the beach or shingle structure is above normal tidal levels and most storm waves. Taken together these provide a very effective barrier to incursions by seawater discussed above. However, despite this the spectre of eroding, or changing beach levels has resulted in many instances of artificial protection measures including building structures on the beach and offshore, re-profiling the beach and beach nourishment.

Shingle beaches by their nature tend to be robust and able to withstand sometimes massive and rapid change. They are particularly well adapted to changing environmental conditions responding to tides, waves and storms by absorbing their energy. The movement of shingle, which often accompanies particularly violent storm events, provides the means of adaptation. Thus natural beaches tend to be lowered and flattened by 'over-washing' as storm wave energy is dissipated (compare the 'natural' and 'engineered' beach profiles in Figure 03-01 below). Although more frequent overtopping and flooding can occur the violence and severity of these events is lessoned. Anyone who watches storm waves approaching a wide sandy beach or a shingle foreshore should need little convincing of their ability to withstand and dampen wave energy. At the same time the movement of material up the beach helps to rebuild the crest height and with it the ability to withstand the next storm.

Thus shingle beaches by responding to storms, and over the longer term to sea level change, by migrating alongshore or landwards can provide an effective and sustainable coastal defence. The issues surrounding this are discussed in more detail in relation to decisions taken at Cley-Salthouse.

Here reprofiling of the area shown in Figure 03-01a will cease and allow a more natural beach profile to develop (Annex 07 and 07a).



Figure 03-01a An artificially reprofiled beach at Cley North Norfolk, October 2001. The picture was taken looking southeast along the coast.



Figure 03-01b. A 'natural' beach profile Cley, North Norfolk, October 2001. The picture was taken looking northwest from the same spot as Figure 03-01a above.

Cheniers as natural sea defence structures

Some areas of the Essex salt marsh coastline are currently protected to some degree by the presence of shell and shingle ridges called 'cheniers'. In Great Britain these features are largely confined to Essex. The ridges are transgressive, lying on a marsh surface, over which they are retreating, isolated by mudflat or marsh to seaward. They occur from Canvey Island to Harwich and vary in their composition from almost pure *Cerastoderma edule* shells in the south to mixed sand, shingle and shell further north.



Figure 03-02 Shell shingle forming a natural sea defence off Cuddy Point, Blackwater Estuary, Essex

Richards & Pye (2001) are currently investigating the composition of the Essex cheniers in relation to their effectiveness as 'Natural Dynamic Coastal Defences'. Once the sedimentary and morphodynamic characteristics of different cheniers are better known, it is hoped to use this information in the construction of artificial cheniers at 'at risk' sites.

Sediment budgets and coastal defence

An important point to consider in relation to coastal defence is that many of the beach-barrier systems on the modern coast were initiated during the Holocene marine transgression and are now responding to changes in sea level, storms and tidal movement involving the reworking of an essentially finite sediment supply (Pye 2001). This poses the most significant management issue in relation to the sustainability of coastal shingle beaches and structures, especially for sea defence purposes.

The situation in Start Bay (Slapton, Annex 02) is summarised in the Futurecoast project as follows:

"There are no significant contemporary sources of material to this coastline. Landsliding in the vicinity of Brixham can occur in response to easterly storms and the removal of toe debris providing a local, although not significant, source of beach material. Toe-trimmed headland slopes fringed by some raised beach deposits are also potential sources in the north, but they are not presently significant. **This indicates that the sediments present in Start Bay were derived from the seabed during the Holocene period**. This is consistent with the high proportion of flint pebbles (75%), present on the barrier beaches, which have no local source." (Halcrow 2003).

A project funded under Interreg II (Chapter 10) with matching funds coming from the relevant local authorities aimed to investigate the supply and depletion of shingle on 'Rives Manche' beaches under present day management regimes, and to determine the sustainability of the beaches in a period of rising sea level (http://www.geog.sussex.ac.uk/BERM/index.html). The study **BERM** (Beach Erosion in the Rives Manche) concluded that overall flint production on the English side of the Channel amounts to only 4,610m³ per annum and may reflect the relatively low cliff heights and the lower flint content of the chalk (as compared to the French coast where the parallel study took place). The major source of flint on the English side is located between Cuckmere Haven and Birling Gap, which is towards the east. Limited observations from qualitative air photo interpretation suggest that beaches bound on the updrift side by engineering structures (such as the Seaford breakwater) have gained in

volume over the past 26 years. However cliff fall analyses along the French coast suggest an increase in the number and frequency of cliff falls since the 1970s following the construction of groynes and jetties and associated reductions in shingle movement.

Comment: *It is not clear from this study, therefore, whether building structures to arrest longshore drift is beneficial or not, even in areas where new material is being delivered to the system. The increase in cliff erosion on the French coast suggests that a reappraisal of approaches, which seek to arrest shingle movement is needed.*

Other sediment sources may be important. Gemmell et al. (2001) have shown in relation to Spey Bay (Annex 08), for example, that it is vital to regard the coastal and riverine sediment processes as a dynamic continuum. Many previous *ad hoc* attempts to protect land from flooding and erosion have failed because of a lack of understanding of changes in the extent of shingle along the coast. Gemmell's work has shown the need to conserve sediment transport and supply and to allow minimum intervention to type of sediment and rate at which it is supplied to river, coast or downdrift beaches.

Comment: *The situation in Spey Bay may be somewhat unusual in the British context as there is a relatively large volume of new material being fed to the system from the river, unlike those on the south coast of England*.

Groynes and artificial breakwaters

Groynes (Figure 03.03) of many shapes and sizes are specifically built to help hold material on a beach and slow down longshore drift. They can appear to be successful and in some instances are able to support the maintenance of a beach for some years. There are however caveats to their use:

1. storms, which can move material onshore and offshore may cause a rapid depletion of the beach. This may involve the pebbles either being moved offshore into deeper water where they can be effectively 'lost' to the system or onshore as the shingle beach 'rolls-over' the land;
2. as material is held up on one section of the coast it can 'starve' adjacent sections 'down-drift' of the groynes.



Figure 03.03. Wooden groyne creating a build up of shingle on the up-drift (left hand side) with a lower beach on the down-drift (right hand side), Porlock (Annex 01), Somerset.

Artificial breakwaters are normally built offshore and designed to shelter the coast by forcing waves to 'break' and lose much of their energy before they reach the shore.

Beach re-profiling sea defence activities

Reshaping shingle structures in order to enhance their sea defence function is a common activity. Although this alters the nature of beaches and shingle structures it does not cause their destruction. However, as with other forms of sea defence they can reduce the availability of material downdrift potentially exacerbating erosion trends there. At the same time the act of moving the beach material does have a profound impact on the vegetation and any invertebrates associated with stable or semi-stable shingle structures. Problems can occur where long-term works are carried out as this will prevent any natural beach profiles being allowed to develop. In some cases emergency repairs can be even more harmful, affecting beaches that have remained stable for many years as was the case on the south coast at Climping Gap, West Sussex in 1992, adversely affecting the Site of Special Scientific Interest.

The situation at Porlock (Annex 01) in Somerset and the along the North Norfolk coast at Cley-Salthouse (Annex 07) suggest that in some cases re-profiling may not be sustainable in the long term. It is also not clear if such activities may cause sediment loss from the beach in the long term and create a weaker internal structure to the beach. These issues are considered in more detail in the habitat guidance (Chapter 06) and discussed in relation to the pros and cons of undertaking 'shingle beach protection'.

Beach nourishment

Beach nourishment is often seen as a more environmentally friendly 'softer' option for coastal defence. However the import of material foreign to the locality can bring with it its own pressures. Imported marine-dredged shingle can cause nutrient enrichment and adverse effects on local beaches when using shingle of unsuitable type/size.

The use of marine dredged material (see below) can be problematic. Excavation is carried out under licence from the Crown Estate and subject to the 'Government View' procedure (dealt with in Chapter 06 where the implications are considered in relation to marine mineral extraction). Economic factors, technical constraints and the occurrence of suitable deposits of sand and gravel dictate the location of dredging areas. Distance from the licence area to the point of landing and market is also critical in determining the commercial viability and competitiveness of marine aggregates. Thus suitability for use in a particular location may always be a secondary consideration.

Beach recycling

Shingle recycling such as takes place at Dungeness and Rye are less problematic than beach nourishment schemes as the material is almost always derived from the local source and 'recycled' to replenish the system. The scale of these activities on the south coast provides important information on the effects on existing beaches. At Dungeness (Annex 05), longshore drift is from west to east and the tip of the ness would naturally continue to move eastwards. The short-term consequence of this would be a deficit of material immediately in front of the nuclear power stations. Clearly the nuclear power station needs protecting and as a result, beach nourishment is carried out to build up the shore and protect the buildings from being undermined and possibly falling into the sea. This is undertaken by collecting material at the east of the site and moving it to the west. Longshore drift ensures the material is carried to the shoreline in front of the station and eventually back to where it was originally collected in the east. From here it is scooped up again and carried back to the west: a never-ending cycle of recharge. Although this protects the power station it prevents the accretion of 'pioneer' shingle ridges on the east side of the site.

At Rye Harbour the current defences are maintained by recycling shingle that has collected against the River Rother Harbour arm. It is thought that these could be breached as frequently as once in five years without the further beach recharge schemes that are planned or implemented further east. At Church Norton Spit, Pagham Harbour, the seaward side is also replenished, annually by the Environment Agency, who recycle large quantities of shingle from the foreshore over a two-month period in winter, usually January/February (see Annexes 03 & 04).

Offshore aggregate extraction

Gravel deposits suitable for commercial exploitation are relatively restricted around the UK coast. As a result the areas which are exploited are themselves limited. This together with access to ports for offloading the material have led to a concentration of dredging licences in areas such as the Outer Thames Estuary, off Great Yarmouth and around the Isle of Wight, and their absence in others, for example Lyme Bay and the western approaches.

According to the Crown Estate who license extraction these license areas cover a very small % of the total sea bed and even within the licensed areas only 15% is excavated in any one season. The location of the extraction sites is shown in Figure 03.04 below.

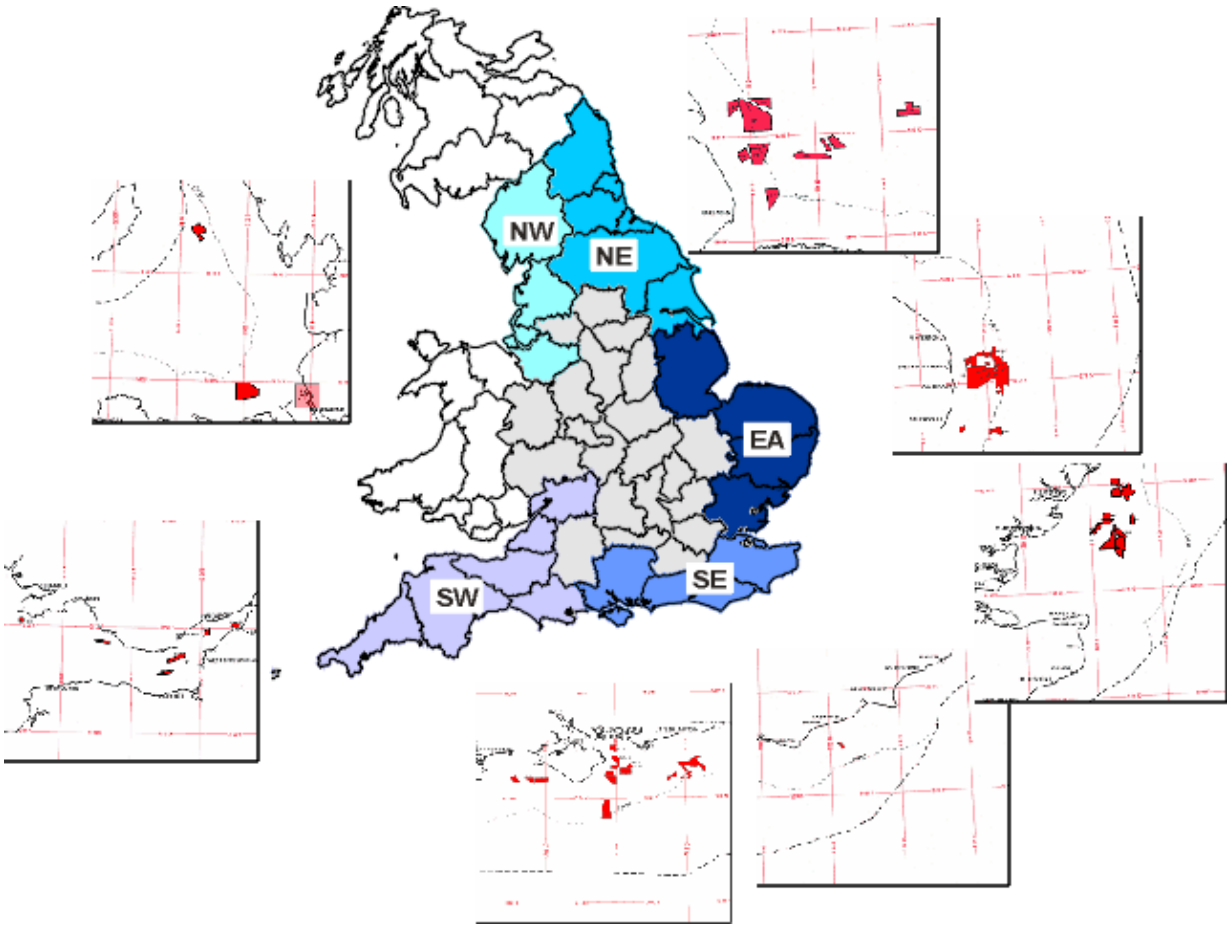


Figure 03.04 Location of sand and gravel licensed areas around the coast of England (DEFRA regions are shown in shades of blue). The extraction area maps are derived from the Crown Estates web site.

The effect of this offshore extraction on the coastal shingle beaches and structures is considered to be minimal as all applications are carefully vetted. Most extraction takes place at a depth of between 10 and 35 metres, though up to 50 metres is possible. At these depths it is generally assumed that there is no impact on adjacent shores. However the process of granting licenses for extraction has become more stringent over the years and Environmental Impact Assessments are now required. Amongst other things this includes the need to assess the physical impact on coastal erosion (through a Coastal Impact Study), see Marine Mineral Guidance Note 1 (MMG 1). Most of the rest of the assessment is concerned with the effects on the seabed and other marine factors.

Offshore dredging - marine aggregate licenses

Marine aggregate extraction has, in the past, been encouraged by Government policy, subject to detailed assessment of potential impacts, known as the “Government View”. Due to the localised occurrence of suitable material and the need for this to be readily exploitable and near to discharge

ports the areas of exploitation are relatively small. The current procedure within each of the countries is summarised below and an attempt made to identify key elements which are important to coastal vegetated shingle.

England

In England the Crown Estate has been responsible for issuing licences. Since 1968 this has been preceded by a 'Government View' (GV) procedure which has tightened up the control of extraction. This procedure involved an analysis and consultation process co-ordinated by The Department of the Environment, Transport and the Regions (DETR). From 1989 all dredging applications have been screened as to the need for an Environmental Impact Assessment (EIA dealt with in Chapter 05) as part of this 'Government View' process and since 1993 all applications have required an EIA comprising a commissioned Coastal Impact Study and an Environmental Statement.

The Government intends to replace the non-statutory 'GV' procedure with a statutory process, with the introduction of the *Environmental Impact Assessment* and *Habitats (Extraction of Minerals by Marine Dredging) Regulations*. This will transpose the EIA and Habitats Directives into national law in so far as they relate to marine aggregate dredging. In 2002 a policy guidance note was published by The Stationary Office on behalf of the Office of the Deputy Prime Minister (available to download from the ODPM web site under planning policy guidance http://www.planning.odpm.gov.uk/policy.htm).

This guidance document "*Marine Minerals Guidance Note 1 Guidance on the Extraction by Dredging of Sand, Gravel and Other Minerals from the English Seabed*" includes detailed information in Annex A on Environmental Impact Assessment and what it should cover. It applies both to applications for dredging licences made under the 'Government View' procedure and for Dredging Permissions made under a new statutory system when it is introduced. The guide also indicates what should be assessed through the ***Coastal Impact Study*** (see Chapter 06).

Scotland

Although marine dredged sand and gravel has not contributed to the supply of aggregates in Scotland to date, the Scottish Executive recognises that it may do in the future. The National Planning Policy Guidelines 4 (NPPG 4) - Land for Mineral Workings - section *additional policy guidelines for individual minerals* includes a paragraph which states, in relation to marine minerals exploitation that:

"*…the potential to do so exists. Proposals are subject to the 'government view' procedure which is co-ordinated by The Scottish Office Environment Department. If favourable, licences are granted by the Crown Estate. Dredging of sand and gravel may be acceptable, provided it can be done without unacceptable damage to sea fisheries and the marine environment* ***or the stability of the coastline****. The operation of the 'government view procedure' is currently under review*."

Available via the Scottish Executive web site, publications @ http://www.scotland.gov.uk/publications/recent.aspx - search on NPPG 4.

Wales

The National Assembly for Wales (http://www.wales.gov.uk/index.htm) is also currently examining responses to its draft policy framework on marine aggregate dredging.

Impact on the marine environment



A number of other Key Issues and 'good practice' have also been identified which relate more generally to the marine environment (see boxes below). These are taken from the Marine Stewardship Report: "Safeguarding Our Seas: A Strategy for the Conservation and Sustainable Development of our Marine Environment". This report can be downloaded from the DEFRA web site http://www.defra.gov.uk/ under 'Environmental Protection in the Marine Environment').

'**Regional Environmental Assessment**'

The Crown Estate has recently granted prospecting rights in an area of seabed 20 miles south of the Sussex coastline, in the deeper (40 metres plus) waters of the eastern English Channel. Ten planning applications to dredge parts of the area have now been submitted by six companies to the Office of the Deputy Prime Minister. Recognising the need for an overall study of the cumulative impacts of their proposals, the companies involved have commissioned a comprehensive Regional Environmental Assessment (REA) and have joined forces as the East Channel Association to fund it. The REA was published in January 2003. A non-technical summary covering an area within which multiple extraction applications might be made; Eastern English Channel is available @ http://www.eastchannel.info/english/index.htm. This appears to be a new approach which fulfils the Government's requirements for an Environmental Statement whilst at the same time taking the wider sub-regional seas area into account.

Key issues an EIA needs to address:

- the coast and coastal processes;
- sediment transport pathways;
- marine ecosystems;
- commercial fisheries;
- water quality;
- navigation;
- marine conservation areas;
- archaeological sites;
- other uses of the sea.

Good practice in aggregate extraction will protect the marine environment through:

- the careful location of new dredging areas;
- considering new applications for Dredging Permissions in relation to the findings of a full EIA;
- minimising the overall impact of dredging by reducing the risk of cumulative impacts from multiple dredging operations and other human activities, minimising the area being dredged at any one time, and minimising the total area permitted for dredging;
- controlling dredging operations through the use of legally enforceable conditions attached to dredging permissions;
- requiring operators to monitor, as appropriate, the environmental impacts of their activities during and on completion of dredging.

The increasingly important issue for shoreline management is the extent to which offshore extraction potentially impacts on nearshore and coastal processes. This is discussed in more detail and in relation to the procedures identified above, in the guidance given in Chapter 06.

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Chapter 04 Protected sites

This chapter deals with the protection of coastal vegetated shingle sites in Great Britain. In the UK there are a hierarchy of designations, which are applied to sites deemed by the statutory countryside agencies to be important for their wildlife (flora and fauna), geological or physiographical features. These include:

Site of Special Scientific Interest (SSSI)

The Wildlife and Countryside Act 1981 (as amended by the Countryside and Rights of Way Act 2000) is the principle legislation under which actions affecting the wildlife interests of designated shingle sites are controlled. It is under this Act that Sites of Special Scientific Interest are designated. Sites are assessed, identified and designated by English Nature, the relevant countryside agency in England (Countryside Council for Wales; Scottish Natural Heritage).

Notification as a SSSI is primarily a legal mechanism to protect sites that are of particular conservation interest because of the wildlife they support, or because of the geological features that are found there. SSSIs can be found all over Great Britain and help to form a national network of areas with the greatest value to wildlife or geological conservation.

The legislation was further strengthened by the Countryside and Rights of Way Act 2000 which amends the 1981 Act provisions and improves protection for SSSIs in England and Wales.

[English Nature has produced an excellent booklet explaining the role of SSSIs and the obligations on owners and occupiers and on the statutory bodies. It includes supporting information on more technical aspects of the process and can be downloaded from the EN web site @ http://www.english-nature.gov.uk.] Details of each SSSI in England can also be obtained from the English Nature web site or via the Department of Environment, Food and Rural Affairs 'Magic' @ http://www.magic.gov.uk/, for most sites pdf files provide the SSSI description (Figure 04-01a shows the location of important sites in England).

Similar information will be available on the Countryside Council for Wales web site during 2003 see http://www.ccw.gov.uk/ (Figure 04-01b shows the location of important sites in Wales). Information for sites in Scotland should be available via the Scottish Natural Heritage web site http://www.snh.org.uk/; though at the time of writing this was inaccessible (Figure 04-01c shows the location of important sites in Scotland).

National Nature Reserve

National Nature Reserves (NNRs) are established to protect the most important areas of wildlife habitat and geological formations in Britain, and as places for scientific research. NNRs are 'nationally important' and among the best examples of a particular habitat. They are either owned or controlled by English Nature (CCW in Wales, SNH in Scotland) or held by approved bodies such as Wildlife Trusts. The management of these sites has as a first priority the conservation of those natural or semi-natural features for which the site has been designated.

Local Nature Reserve

Local Nature Reserves (or LNRs) are designated under Section 21 of the National Parks and Access to the Countryside Act 1949 by principal local authorities (county and district councils). Parish and town councils can also declare LNRs but they must have the powers to do so delegated to them by the principal local authority.

Natura 2000 sites

At the centre of the European Union policy is Natura 2000, a coherent network of protected areas across the EU. These areas are made up of Special Areas of Conservation (SACs) and Special Protection Areas (SPAs). [The location of the candidate SACs and SPAs in Great Britain are shown in the maps below (Figure 04-01a, b c). A description of the interest, as provided by the JNCC web site is reproduced in Appendix 02 for each of the SACs.]

An overview of legislative implications of these designations and the various regulatory controls that may affect proposals to manage or restore coastal vegetated shingle habitats is provided in Appendix 01.

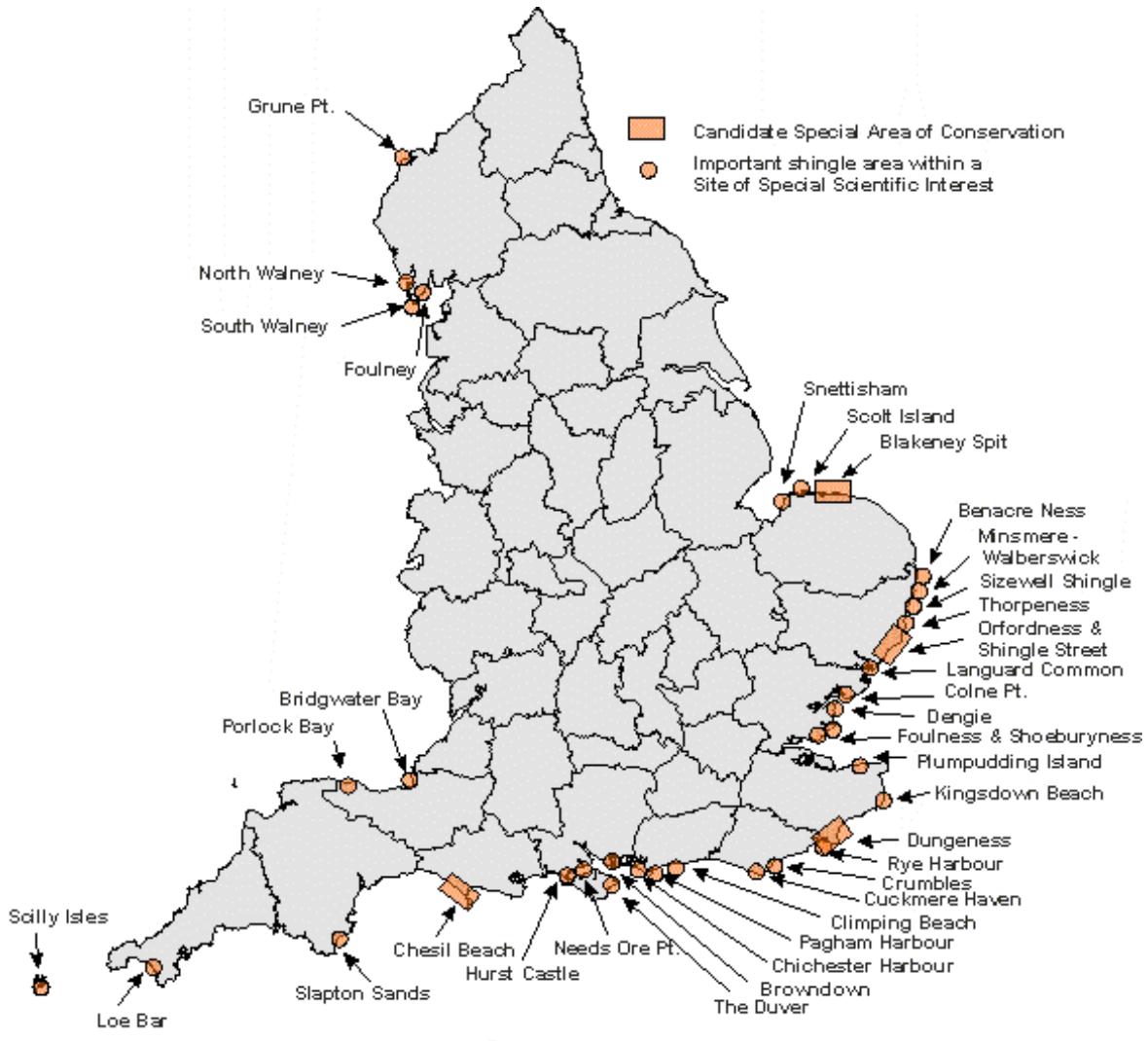


Figure 04-01a Important shingle sites in England. NB The Crumbles is the only site mentioned which is not a Site of Special Scientific Interest

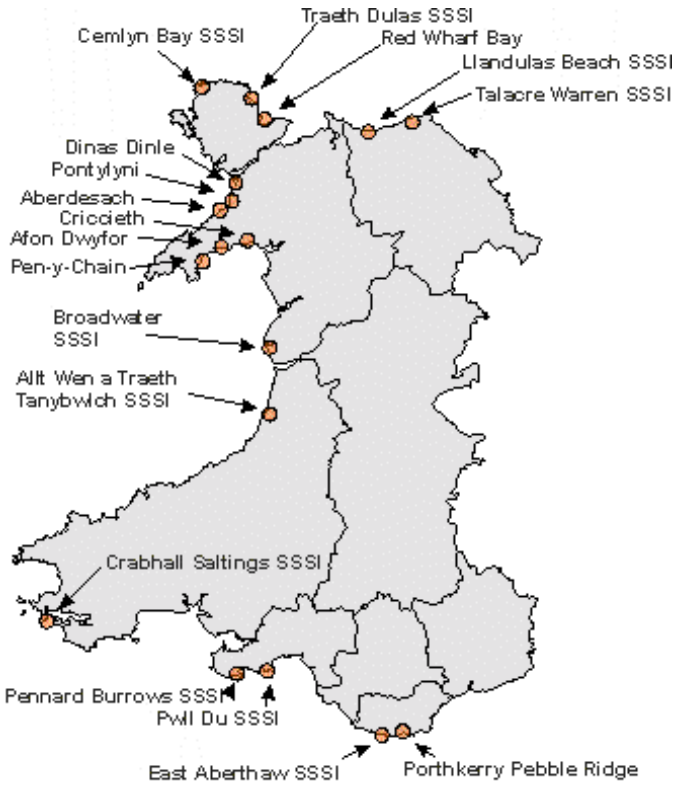


Figure 04-01b Important vegetated shingle sites in Wales

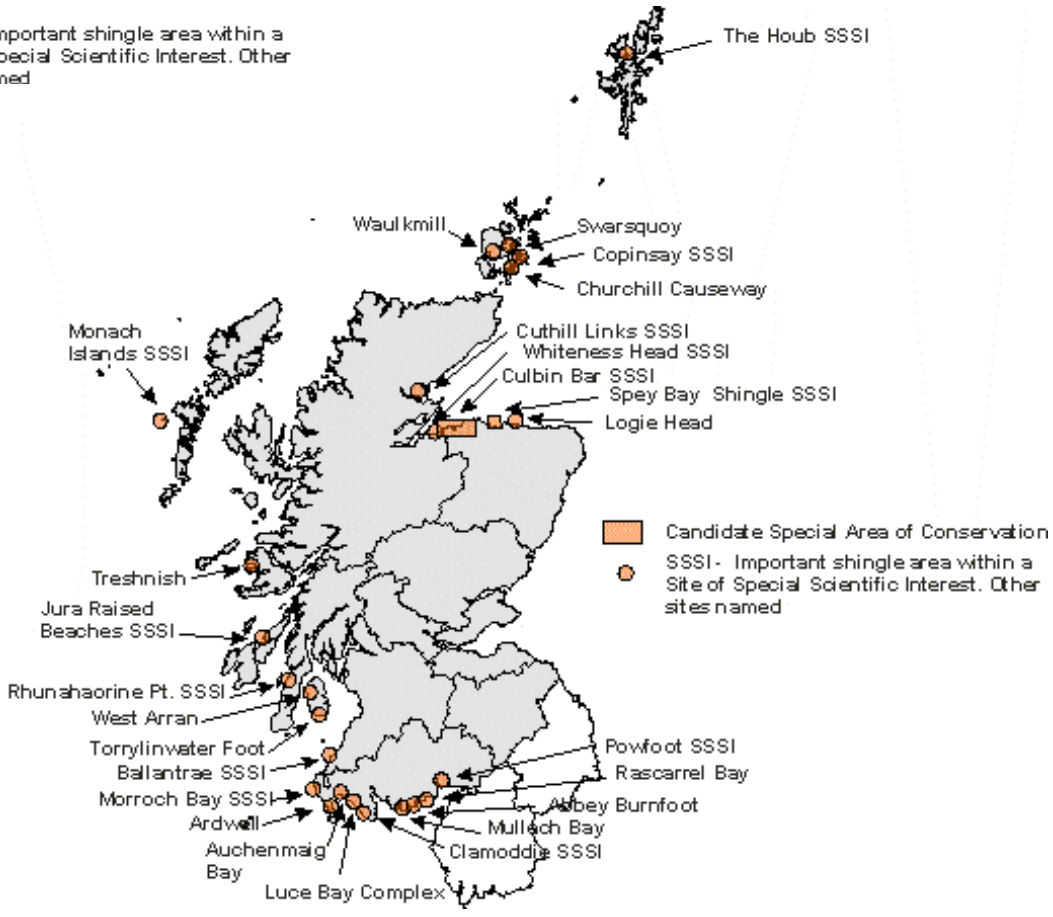


Figure 04-01c Important vegetated shingle sites in Scotland

Chapter 05 Pressures and threats

Introduction

Shingle beaches by their nature are ephemeral and prone to sometimes massive and rapid change. As has been described in Chapter 02 the plants and animals which survive there are therefore usually tolerant of periodic disturbance. However, once the ridges become stabilised and out of reach of storm waves, a gradual build-up of interstitial sediment takes place and with it the development of more mature and stable vegetation. At this stage the communities, which become established are adapted to highly stressful conditions involving lack of water and substantial temperature fluctuations not dissimilar to those of some deserts. Although such species are highly tolerant of such conditions they are much more sensitive to disturbance.

The most widespread and damaging activity is the removal of gravel either from the beach or more significantly the structure itself. Gravel extraction destroys the vegetation and associated fauna which, due to the complex structure of the matrix in which these communities develop, is impossible to re-create. Infrastructure development is equally damaging and the simple act of driving a vehicle across a series of mature shingle ridges can cause damage which remains visible for many years until the next 'natural' change takes place and the structure is reformed.

The Habitat Action Plan summarises the problems facing the conservation of this priority habitat under the European Union Habitats and Species Directive as follows:

"2.3 Exploitation. *Shingle structures have been regarded as a convenient source of aggregates, and have been subject to varying degrees of extraction resulting in severe alteration of morphology and vegetation (e.g. Dungeness and Spey Bay) or almost total destruction of major parts of the feature (e.g. Rye Harbour). Industrial plant, defence infrastructure and even housing have been built on shingle structures (e.g. Dungeness, Orfordness, Spey Bay), destroying vegetation and ridge morphology. At Dungeness water is abstracted from the groundwater system; there is some evidence of drought stress on the vegetation, but it is difficult to distinguish the effects of water abstraction from those of gravel extraction*."

Thus there are a number of key areas where pressures on shingle beaches and structures have caused significant, direct and irreversible loss of shingle habitat (nearly 100% in the case of the Crumbles, see Figure 05-02 below and more than 40% of the surface of Dungeness, Annex 05). These losses and the impact of activities associated with coastal defence have had a major impact on the habitats and led in part to the need for this guidance manual. Other activities alter, rather than destroy the habitat and some of these changes can be reversed, offering opportunities for habitat restoration. A summary of the most significant issues is given below:

1. aggregate excavation both:
    - onshore;
    - offshore [**Dealt with in Chapter 03**].
2. habitat loss due to structures built on the surface shingle including:
    - infrastructure development, industry, housing and roads;
    - power stations including nuclear facilities;
    - military use, car parks, airfields;
3. sea defence and coastal protection [**Dealt with in Chapter 03**] involves creating:
    - groynes and artificial breakwaters;
    - reprofiling fringing beaches;
    - beach nourishment.

4. recreational use involving:
    - trampling, destroying vegetation and ground-nesting birds;
    - disturbance of nesting birds;
    - boat mooring;
    - fishing;
    - vehicle access.
5. other issues:
    - grazing (loss of rare plants);
    - water abstraction;
    - military training;

Habitat loss - excavation

Gravel is a rare material highly desired by the construction industry. As a result, large quantities are obtained by extraction from both onshore and offshore deposits. The former provide a ready source of easily accessible material but can cause long term damage to the surface shingle. The latter is more difficult to exploit and impacts on the marine environment can also be significant. Knock-on effects on ,in particular, coastal shingle beaches and structures should also be considered.

Onshore excavation

The most widespread and damaging activity is the removal of gravel either from the beach or more significantly from within the body of the shingle structure itself. Deep excavations create a void which can rapidly become filled with water, creating gravel pits (Figure 05-01). Several large sites have been adversely affected in this way. Dungeness, the largest shingle structure in Great Britain covering some 1600 ha of exposed surface, is the most disturbed site with some 43% of its surface affected by a wide variety of adverse impacts. These range from relatively minor incursions into the intact vegetated shingle ridges by vehicles (which break up the vegetation), to the complete destruction of the vegetation and ridge structure through the extraction of gravel (Fuller 1985).



Figure 05.01 Dungeness, Kent. Gravel extraction and the resulting gravel pit in the body of the shingle structure. Note the nuclear power stations in the background.

A similar and even more complete loss has overtaken the Crumbles in southern England where a 160 ha site, immediately east of Eastbourne, has been destroyed. Building works, including erection of houses and gravel extraction have destroyed most of the shingle surface and the remaining vegetation on intact ridges has been broken up by heavy visitor pressure. Figure 05-02 depicts the situation at Pevensey Bay Shingle (the Crumbles), southeast England, showing damage to shingle ridges from buildings, gravel extraction and recreation.

Extraction may not always be so destructive. Shallower excavations, which do not reach the water table can, if left undisturbed become recolonised. In some cases this may increase local biodiversity of

the site, as species associated with damp to wet conditions become established (see parts of Dungeness and the coastal river shingles at Kingston, Scotland, Annex 08). Such changes, however, must be viewed in the light of the fact that coastal vegetated shingle cannot accommodate further losses, without significant impacts to its overall extent and distribution.

Shingle beaches may also suffer loss through localised excavation. This may be by individuals who remove small quantities for personal use or larger scale operations for sale. It has not been possible to find information on the extent of these activities, which are both illegal, thus assessing their impact has not been possible. Whatever the cause the removal of surface shingle destroys the vegetation and associated fauna, further degrading the habitat.



Figure 05-02 Pevensey shingles (the Crumbles) showing the extent of the destruction of the surface shingle. The limit of the original shingle structure which has been eroded over many years. Another example of a 'coastal squeeze'.

A similar story can be told for many other areas. For example, much of the surface shingle west of Rye Harbour, again in southern England, has also been destroyed by gravel extraction. Kingston shingle, which lies to the west of the River Spey in Scotland, is another important area where shingle extraction destroyed much of the ridge structure, probably during the 1940s (Annex 08). Though less information is available French sites have been similarly affected "The small sand and gravel ridges of Basse Normandie (south of the Seine) have in large part been urbanised" (Géhu & Géhu-Franck, 1979). Other sites in France such as Cayeux sur Mer (Côte d'Opale), Sillon de Talbert (one of three locations on the north Brest peninsula) and two more to the south have also been degraded and diminished from gravel extraction.

Offshore aggregate extraction

Offshore aggregate extraction is not an obvious and direct pressure on coastal shingle formations. However the extent of gravel extraction offshore poses a potential threat if it diminishes the supply of material onshore. The precise effects are difficult to determine. The issue has been covered in some detail in Chapter 03, sea defence and coastal protection issues.

Habitat loss to infrastructure

Shingle structures have suffered much damage both in the UK and abroad from direct destruction of the surface shingle. Infrastructure development includes buildings such as housing, recreational facilities, access roads and railways and in a few instances nuclear power stations (Dungeness) and military buildings (Orfordness). Even the simple act of driving a vehicle across a series of mature shingle ridges will cause lasting damage, which remains visible for many years. Several large sites have been adversely affected by gravel extraction and other damaging activities; these are included in the site studies annexes.

Industry, housing and roads

Major industry is not normally built on shingle. The area of land is often too small or too remote. However there are numerous examples of housing and other buildings built on the more stable upper shores as in Pevensey Bay, East Sussex; Shingle Street, Suffolk and Lydd-on-Sea, Kent (Figure 05.03). Other areas may be built on shingle such as the town of Brighton (where there is a substantial shingle beach) but it is not clear the extent to which these have resulted in major losses of habitat. At Shoreham Beach a village was developed for the film industry on the spit in the early 20th century. This has now expanded to take over much of the shingle spit other than the foreshore. At Pontlyfni, Gwynnedd, much of the northern part of a high quality shingle beach has been lost to the construction of a sewage works (Randall, Sneddon and Doody 1990).



Figure 05.03 Lydd-on-Sea, Kent. Housing and a road interrupting the transition from shingle beach to more stable vegetated shingle ridges. Note the parallel ridges on the right and *Crambe maritima* the large glaucus plant in the foreground.

Power stations including nuclear facilities

The need for a coastal location for power stations, providing access to cooling water, has led to the construction of some substantial facilities on shingle. The most notable in the UK is the nuclear power station complex on Dungeness, although there are other examples of development on or near other coastal and some shingle sites (e.g. Sizewell, Suffolk, East Aberthaw in South Wales and Hinkley Point, Somerset). Whilst power station construction leads to an obvious loss of the surface shingle, such facilities can also result in additional damage. This is particularly true at Dungeness where access roads, power lines, vehicle access associated with their maintenance and ironically attempts to improve the biodiversity of the adjacent shingle habitat have all caused loss of, or damage to the shingle surface and its wildlife.

A secondary and in some ways more important consequence of building a nuclear power station on shingle, lies in the need to retain buildings in situ for many years. In the case of Dungeness this is particularly important as it has been built in an area where the shingle beach is attempting to move eastwards as it has done for centuries. This has resulted in the need for a continuing cycle of beach nourishment with material being moved from the beach to the east of the building and transported to the west where it is deposited back on the beach. The natural process of longshore drift then carries the material back to the east. [Annex 05, Dungeness]. This process also prevents anything other than the rudimentary foreshore shingle communities from developing because of the continued disturbance of the shingle. In this sense it is similar in effect, to beach nourishment and reprofiling.

Military use, car parks and airfields

Most shingle structures are not big enough to accommodate major military use for the provision of airfields; the exceptions are Orfordness and Dungeness. At Orfordness military activity has destroyed, or seriously disturbed a major part of the site (see site issues report). At Dungeness both military use and an airfield have also affected the area of surface shingle (see site issues report). At many other sites (unquantified) formal and informal parking has caused loss or compaction of the surface shingle. This is especially the case at many of the shingle beaches in Scotland. This problem is examined in detail for the Solway coast by Randall (2001). The reports prepared by Sneddon & Randall (1993 a, b,

and 1994 a, b) also contain a considerable amount of information about pressures on the sites included in their study.

Sea defence and coastal protection

The construction of coastal defences can have potentially serious implications for shingle habitats. In summary defence works may result in changes to the movement of material both on and offshore and alongshore. This has important consequences for the natural dynamics of the habitat. The structures themselves can also lead to a loss of overall extent in habitat. Beach nourishment, feeding and re-profiling are also significant activities, the former having both positive and negative impacts on the habitat, the latter almost always having negative effects. Beach recycling represents an ongoing negative impact on many designated sites. (The issues surrounding these activities are discussed in some detail in Chapter 03 on flood and coastal defence.)

Recreational use

Recreational activities can affect both shingle beaches and structures. In places in Baltic Germany and Norway extensive 'flintstone fields' occur extending inland to *Calluna* heathland and birch/juniper scrub similar to those in Kingston, Scotland. Several of these have suffered severe recreational damage. On the most mature structures even relatively small incursions into the surface layer may remain visible for many years. The impact of off-road vehicles ranges from relatively minor incursions into the intact vegetated shingle ridges, which break up the vegetation, to prolonged use and total destruction. Wheel tracks created by vehicles during the 1940's at Dungeness, for example, are clearly visible today. Many of these actions can be cumulative leading to long term and irreparable damage to the shingle surface. Both recreational activities may also have an impact on other important features such as bird nesting survival and behaviour.

Several Scottish sites, including Rhunahaorine, west Arran and Auchenmalg have official camping and caravanning sites either on, or immediately adjacent to the shingle. Many of these sites are located on raised beaches or forelands and therefore damage may occur on the more terrestrial parts of a site.

Vehicle access

As already indicated above, driving vehicles onto shingle causes considerable disruption of the surface. Vehicles can create swaths of damaged and destroyed vegetation. The structure of the shingle may be completely destroyed and its ability to hold water and hence support plant and associated animal life. Elsewhere, beaches and smaller sites can be greatly affected by uncontrolled vehicle access. Even where the surface remains intact, compaction can lead to changes in drainage and plant community composition. Sites on the west coast of Scotland, for example, are often readily accessible from the road and as a result have suffered from vehicular damage. This takes the form of track marks across the sites with destruction of vegetation on areas which are habitually used for parking.

Trampling

Some of the most susceptible communities on shingle structures are those with abundant lichens. These slow growing species are very sensitive, especially in dry weather, when they are particularly easy to damage. Even the passage of one person walking (or cycling) on an established vegetated shingle ridge can leave 'footprints', which may never be repaired. Compaction of the surface may also affect the seed bank making it more difficult for some species to germinate. Control of visitors at Orfordness has helped to rehabilitate communities of the rare *Lathyrus japonicus* (Annex 06).

Disturbance of nesting birds

In areas where ground-nesting birds, including terns occur, then potential damage to nests and their eggs is obvious. Even where direct destruction of nests does not take place repeated disturbance may discourage or displace nesting species. As several of these (notably the terns) are rare and protected and depend on isolated sites where their eggs blend in with the shingle substrate, impacts can be severe.

Boat mooring (including fishing)

Shingle beaches are popular areas for providing access for small fishing boats. Pulling boats up a beach, as frequently occurs on the south coast can lead to loss of vegetation or surface damage. The process of hauling the boats onshore, whether by fixed winches or tractors can cause localised damage to the shingle, which can vary in severity depending on the size of vessels and the intensity of use. Rhunahaorine has a fish farm on the northerly part of the foreland, which creates pressure on that site.

Other issues

Grazing (loss of rare plants and invertebrates)

Grazing of domestic stock is a relatively restricted activity even on the larger shingle sites. The sparseness of the vegetation and limited growth rates combine to make the available herbage limited and unpalatable. Hares may frequent these areas and are especially plentiful on Orfordness. Only in Scotland where sheep and cattle from adjacent areas can have access to beaches are they likely to have adverse effects - in this case the loss of *Mertensia maritima* a rare plant of beach shingle in the north. (Figure 05-04, Randall 1988).



At Rhunahaorine Point, western Scotland heavy grazing of much of the foreland by cattle and sheep with relatively high stocking levels has reduced the area of heath. This is not only because of the direct effect of grazing on the vegetation but also the indirect effect of the fertilisation it supplies to an otherwise nutrient-poor environment. Rhunahaorine also has a fish farm on the more northerly part of the foreland. Grazing pressures can also reduce structure and diversity of the vegetation and with it losses of invertebrate fauna.

Figure 05-04 Cattle grazing *Mertensia maritima* Treshnish beach Isle of Mull

Arable cultivation

At several raised shingle beach sites in Scotland the rear part of the beach has been reclaimed for arable agriculture with the total loss of topographical features and the more terrestrial components of the vegetation. A classic example of this is the site at Claymoddie (Figure 05-05 below), which is otherwise one of the most undisturbed along the Solway Firth (Randall 2001). Much of the land on the lower, younger ridges at Rhunahaorine has been reseeded with agricultural leys, which has resulted in a loss of natural vegetation. Arable farming also occurs on the raised gravel beach at Browndown, Hampshire (Randall et al. 1996).

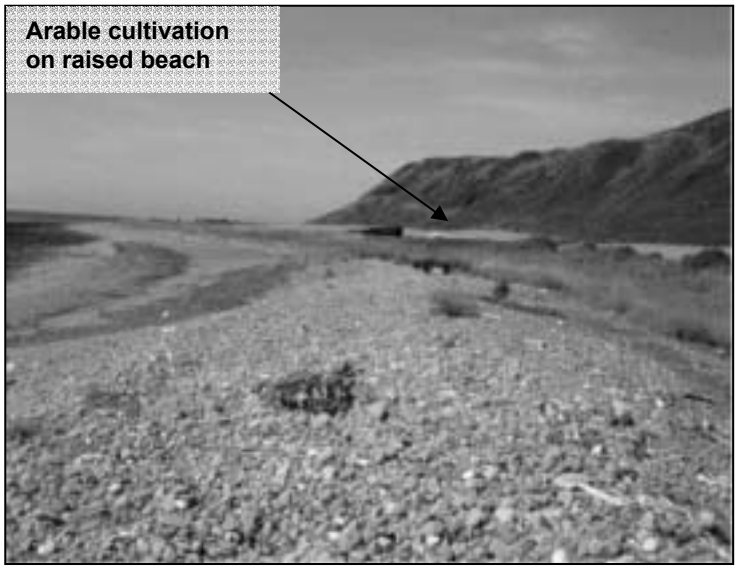


Figure 05-05 Loss of shingle to arable cultivation at Claymoddie, Dumfries and Galloway

Plantations

The major loss of natural vegetation on the shingle at Kingston Shingles has resulted from the planting of pine on the western part of the site (Annex 08). This has not only caused a direct loss of native vegetation but also has major implications for species composition in undisturbed areas where seedlings of *Pinus sylvestris* have established. A similar problem is occurring on Culbin Bar where seeding *P. sylvestris* have spread from the plantation on the mainland (Annex 09).

Water abstraction and reservoir use

Water abstraction can lead to lowering of the water table and saline water intrusion. This is an issue at only one site, Dungeness where water is used for domestic purposes. The shingle aquifer here was first used to supply water to Littlestone-on-Sea at the end of the nineteenth century. Abstraction has continued ever since and rates have increased substantially in the last 50 years leading to some problems with saline water intrusion (Annex 05 and Chapter 02, section on hydrology, page 13).

Dumping of waste

Because the value of shingle areas is not always apparent to some members of the public, they are frequently used as dumping grounds for rubbish. Kingston Shingles, Grampian has been used as a tyre dump and Philip and Mary, Dumfries and Galloway has been severely damaged by the dumping of waste road metal. In several sites such as Ballantrae, Strathclyde and Shore Cottage, Dumfries and Galloway the dumping of farm organic waste has caused local eutrophication.

Military training

The direct effects of habitat loss due to military infrastructure are visible at Dungeness (Annex 05) and Orfordness (Annex 06). In at least one part of Dungeness, the undulating nature of the ridges was considered an obstacle to military training. These were flattened and any vegetation destroyed. The presence of a 'holly wood' was respected and individual bushes now survive on a flat shingle plain (Figure 05.06). The Training Camp at Browndown has also witnessed significant damage to the shingle vegetation, but sympathetic management by the MOD in conjunction with the Browndown Conservation Group has resulted in a more careful policy being adopted (Randall et al. 1996). The fragile lichen communities are readily damaged by military use and recovery can be very slow an important consideration.



Figure 05.06. The 'holly wood', Dungeness, Kent. Note the bare, flat nature of the intervening shingle (and the 'wind-pruning' of the holly bushes).

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Chapter 06 Habitat guidance (management and sea defence issues)

This chapter is concerned with two areas of guidance:

1. management of vegetation in a more traditional sense including the control of invasive species, grazing etc. and the adverse effects of human use, notably for recreation;
2. the guide identifies problems and opportunities associated with sea defence maintenance including recharge schemes.

Before considering any guidance on habitat management or restoration a primary concern must be the conservation of the remaining areas of relatively intact and undisturbed areas of vegetated shingle. All damaging activities must be prohibited on all sites where important areas of this extremely rare habitat exist. The UK has a particularly important role to play here by virtue of the extent of the shingle (gravel) banks that occur here and their world significance (Chapter 01).

Management, including vegetation management

Before discussing the management of shingle in relation to sea defence operations, a more general discussion of management is appropriate. Many of the plant communities supported on shingle are unique to this substrate and this must be taken into account for the management of vegetated shingle. In particular whilst pioneer communities of vegetated shingle can begin to recover naturally from damage within a few years, as long as the seed bank remains intact and further damaging activities are halted, more mature communities will not.

The development of unusual bryophyte - and lichen-rich communities, unique to shingle (Lambley & Hodgetts 2001) has high conversation value. These and other closed-turf communities are unlikely to recover quickly as the lack of soil development, on many shingle substrates mean the vegetation requires many decades to reach maturity. Some can take hundreds of years to establish. It is important that these types of vegetated shingle communities are not damaged, as they are extremely rare.

Traditional forms of management

Varying degrees of management intervention are required to conserve differing successional stages of shingle structures. Though unlike other coastal habitats (notably saltmarshes, sand dunes and sea cliffs) where grazing provides a significant determinant of the plant and animal communities, which develop; for vegetated shingle this influence is relatively limited. Minimum management is required to maintain scrub or woodland communities on shingle but heath, grassland and pioneer communities may require more active management. The presence of lichen heath is indicative of remote areas with limited public access (e.g. Orfordness, Annex 06; lee slopes of Chesil Beach). To allow the development of lichen or moss-rich heath, access should be restricted to such areas, thus protecting the fragile structure of these communities. *Calluna vulgaris* heath on the other hand may require low-level grazing management to maintain age and species diversity.

Edaphic and environmental conditions peculiar to shingle seem to serve as natural limits on cover for *Calluna* (and other plant communities). Grazing management may be all that is needed to restrict scrub invasion on grassland and to a lesser extent heathland. However, following cessation of grazing pressure, maintenance of a variety of grassland (and a few heathland) communities may require clearance of invasive scrub and heath. Reintroduction of grazing may then be required to maintain certain grassland assemblages such as the herb-rich *Holcus lanatus* grasslands of Arran. It must be remembered that, away from the drift-line, shingle structures are usually very nutrient-poor so that the introduction of grazing herbivores on traditionally ungrazed sites will almost always cause vegetation change both as a result of structural effects and nutrient enrichment and is potentially damaging (see for example Rye Harbour, Annex 04).

Soil conservation on shingle is also highly significant. Mechanical clearance of scrub invasion as has been carried out at Slapton (Annex 02) and Rye Harbour (Annex 04) will bring about soil removal and a competitive advantage for ruderal species rather than earlier stages of shingle succession. Because of the long timescale of soil development on shingle, fire, whether accidental or deliberate can affect the full depth of the soil including any seedbank. Locally, ground nesting birds (notably gulls) can also

influence the vegetation composition by increasing levels of nitrogen in what is a nutrient poor substrate (see for example Annex 06, Orfordness).

Summary guidance - traditional management

Traditional management notably by grazing animals is a requirement for the maintenance of some more mature grasslands and heath. However, because of the low productivity of these areas introduction of grazing where none has occurred traditionally should be avoided. Limited intervention by way of scrub control may be appropriate in a few cases.

Control of visitor and other pressures

It almost goes without saying that the first requirement in the conservation of any shingle beach or structure must lie in its protection from damaging work. This will include the building of any structure which destroys and covers the surface shingle. Equally damaging is gravel extraction (see for example Dungeness, Annex 05). On top of these activities further pressure is put on areas of shingle by public use. Trampling, in particular, can compact the fragile substrate and damage the vegetation. Access to certain sites should be restricted to boardwalks rather than footpaths. All of the activities causing loss or damage to the surface shingle should be avoided as repair and restoration is often difficult. Solving these problems involves a combination of physical control measures and education.

Public access

The fragile nature of the shingle habitat makes it important that the public are directed in their movement as much as possible. Public access maps should be produced and explanations given why general access is not recommended. One successful way of limiting general public access is to provide an information board and a nature trail from the nearest parking site. Public access is limited to a certain extent by car-parking provision but in some areas off-road cycling and the illegal use of motorbikes and four-wheel drive vehicles over vulnerable areas of vegetated shingle may cause irreparable damage. Where this occurs this can be prevented by forging links with the local police and where physically possible, such as at Orfordness (Annex 06), access points can be blocked. Once the damaging activity is prevented, successful regeneration of rare plants can take place. However, in areas where access needs to be provided for emergency coastal defence works, it can be difficult to restrict access.

Trampling

Because of the succulent nature of many of the pioneer plants they are particularly susceptible to trampling. Well situated boardwalks across the backshore to the beach front are an efficient means of reducing this problem, as proposed for Dungeness (Annex 05). Locations where this has been successfully carried out are Walmer, Kent and Portland Harbour at the eastern end of Chesil Beach cSAC.

Off-road parking/unauthorised dumping

At Dungeness the judicious positioning of small ‘grips’ or dykes by the roadside or placing bollards protects rare plants growing near the road. Along the Luce Bay shingle, in Scotland, designated access points for parking have been improved while other possible vehicle access points have been made more difficult to use.

Reducing recreational pressures

Trampling disturbance is significant on shingle due to recreational pressure; sometimes grazing animals can also influence the substrate in this way (e.g. Randall 1977). This is particularly true among pioneer and lichen-rich communities which have a very long recovery time. Regional distribution patterns exhibited by some shingle communities highlight the importance of regional

management policies for this habitat. In Britain the Moray Firth emerges as a particularly distinct floristic unit, as do the pioneer communities of the south east which are threatened by recreational pressure.

Signage

Signage and information boards such as those employed at Pagham and Rye Harbours (Annexes 03 & 04 respectively) are a significant element in helping to control damaging activities on sites used extensively by the public. Whilst this may not stop damage, especially that undertaken as a wilful act of vandalism it does go a long way to alleviate some of the worst aspects of damage. When coupled with educational activities a body of knowledge and respect for the areas of interest can be built up.

Environmental education strategy

Shingle sites frequently have a 'wow' factor because of the dynamic nature of the site, the classic nature of the geomorphology and the unusual vegetation communities and wildlife that are present. They are important locations both for formal education (school visits) and for outreach to the general public. Where possible leaflets and interpretation boards should be produced and links made into local and regional networks.

The Nature Coast Project and East Sussex Coastal Biodiversity Project have produced an awareness raising/educational video on vegetated shingle, entitled "Life on the Edge - wildlife of a shingle beach". It is only 7 minutes long but shows what an extraordinary and attractive habitat vegetated shingle is. It also shows the problems it faces. This video would be ideal for use at meetings, training courses, presentations etc or as a continuous display in a reception area/ visitor centre etc.

Summary guidance - controlling damaging human activities

These problems can best be solved by education of local people about the wildlife value of shingle sites. The variety of approaches summarised above can be employed in different combinations, at different times and with greater or lesser emphasis. A key element lies in the approaches being as flexible as the habitats and species which they are trying to protect.

Nest disturbance

Temporary fencing

Nesting sites for certain shingle species such as terns and oystercatchers are typically in beach locations much frequented by walkers and their dogs. Work at Rye Harbour LNR (Annex 04), among other sites, has shown the value of temporary fencing in these locations to reduce nest disturbance.

Identification guides

As with plants, a great deal of disturbance is accidental and unintentional, even by scheduled workers in shingle areas. Laminated identification guides of bird species and their eggs/nests can be provided for machinery operatives and others working in 'at risk' areas.

Seasonal wardening

Because of the dynamic nature of tern nesting patterns, it may be necessary to consider temporary wardening of some sites, as occurs at Blakeney Point.

Summary guidance - protecting nesting birds

Simply delimiting an area sensitive to nesting birds can provide the necessary protection in most places and for most of the more sensitive periods of the year. Identification guides and seasonal wardening also play their part.

Pest control

Pest control is a very sensitive issue and what might be considered pests at one site may be harmless or even regarded as an asset at another. Hence any form of control must be carefully researched before implementation, always remembering that there is a legal requirement to control some pest species (Ground Game Act 1880 as amended 1906, Weeds Act 1959). Be aware of Health and Safety and Control of Substances Hazardous to Health Regulations 2002 (COSHH) which require employers to control exposure to hazardous substances to prevent ill health. Pest control should take place in conjunction with Habitat Action Plans, Biodiversity and Species Action Plans.

Control of alien plants

Because shingle structures are usually 'open habitats', it is easy for alien species to invade and establish in those areas where the abiotic environment is less harsh. This is particularly the case on the south and east coasts where houses and gardens abut the shingle, often resulting in enrichment of the habitat and the spread of non-native garden species. This can often be a sensitive issue as local people enjoy the showy species and some such plants can be particularly important for invertebrate species, e.g. red valerian *Centranthus ruber*.

At Pagham LNR (Annex 03) there are plans, in conjunction with English Nature, to determine the most effective way of controlling a particular alien species by establishing trial plots / quadrats to be treated in a variety of ways, including hand pulling, herbicide treatment and incineration. As the work will be carried out adjacent to houses, awareness-raising among the local residents about the necessity for selective vegetation control will form a fundamental part of the project.

Summary guidance - controlling invasive alien plants

The following techniques are suitable:

- *Senecio jacobaea* (Annex 03 Pagham). Pull individuals in May-June or spray clumps with Clopyralid in April & September;
- *Cirsium arvense.* Top in August or spray in May-June with Clopyralid;
- *Urtica dioica.* Strim when required or spray from April with Glyphosate.

Control of animals

Rabbits - Shooting or ferreting both give limited control. Fumigation by trained contractors is recommended so long as no other species are sharing the warrens. An alternative is to fence those areas in need of protection with wire or electric cable (Annex 04, Rye Harbour). Recommended wire is 19 gauge with 31 mm mesh.

Mink, **brown rat** and **fox** may all become pests where ground-nesting birds are important. Mink are best trapped in early spring, foxes shot from August to February and rats poisoned all year round. Contractors should be used and due care taken that other species are not affected.

Corvids can be trapped in the breeding season and shot at other times (Larson traps were used at Pagham Harbour, Annex 03).

Summary guidance - controlling animals

Traditional approaches of shooting, gassing and poisoning may be applied, subject to the normal procedures for undertaking these operations.

Harrowing

Cultivation with tractor and harrow has been used as a technique to knock back vegetation in areas where rampant vegetation is detrimental to ground-nesting bird species (see Annex 04, Rye Harbour). Bare shingle is a favoured nesting habitat for several rare birds, notably terns; it can also host some rare and highly specialised plant communities and their associated invertebrates. Any decision to remove vegetation must be taken in the light of a full understanding of the conservation consequences including the geomorphic implications of removal of sediment from the shingle supply location.

This management technique also highlights another central dilemma for conservation particularly on vegetated shingle sites i.e. **should management be for nesting birds or vegetation**? Although successful in maintaining open areas of shingle it also results in an unnatural appearance to the topography and may seriously disturb plants and invertebrates that are also important to the site.

Summary guidance - maintaining un-vegetated open shingle surfaces

Re-creation of island sites such as Tern Island at Pagham Harbour (Annex 03) can clearly be effective for the protection/relocation of tern nest sites.

Caution: Problems associated with location of the new construction, implications for sediment regime and the balance between the values of these areas for birds as opposed to other interests need to be addressed. NB There was a problem of plant growth on the newly created tern island at Pagham partly due to the importation of shingle, which already included an amount of small fraction material and seed of ruderal species.

Coastal defence

Providing guidance for coastal defence operations presents a complex series of issues. One of the most obvious is the provision for reducing the impact of engineering works on shingle beaches and structures themselves, as well as any vegetation and associated species. Where reprofiling is undertaken on a regular basis this will prevent the establishment and growth of anything other than the most ephemeral of plant communities and the regular movement of heavy machinery can compact the shingle. At the other extreme beach nourishment or the erection of some coast protection structures (groynes and breakwaters) can help to create conditions where more stable forms of shingle develop and with it typical vegetated shingle communities. However, beach nourishment can be detrimental in increasing the amount of fine sediment within the shingle, hence changing the conditions required by pioneer species. Between these two extremes lie a series of options for reducing or avoiding damage and destruction of this rare habitat.

Issues surrounding the implications of offshore gravel extraction (which may be used for beach nourishment) on the stability of the shoreline are considered in Chapter 03. Chapter 03 also discusses the value of shingle beaches and structures as sea defences in their own right. It is this value which leads to an increasing recognition that allowing the sea to reprofile beaches and move structures landward is as effective a sea defence strategy as one which seeks to retain the beach in its current location. The case of Porlock (Annex 01) lends credence to this approach. The issues surrounding the protection of the Slaughden sea defences at Orfordness (Annex 06) and the reappraisal of the Cley-Salthouse scheme (see Annex 07 and the Living with the Sea Executive Summary included as Annex 07a) are important to the wider debate about managed realignment as an appropriate policy response to eroding shingle coastlines.

In the event that it is decided that intervention is required and acceptable on cost-benefit analysis then there are many ways in which the operation of the works can be managed to reduce the environmental impact. These are considered next.

Conservation during coastal defence engineering works

Vegetated shingle can be damaged in a number of ways when coastal defence and other works are being carried out on the beach. It should be noted that damage could be done not only by vehicles and machinery but also by trampling, especially on the older communities.

Damage can be in a variety of forms:

- physical damage to plants;
- physical damage to shingle communities (including pioneer, intermediate, closed-turf, moss and lichen etc);
- disturbance to breeding birds;
- movement and/or compaction of shingle;
- destruction of the seed bank;
- imported marine-dredged shingle causing nutrient enrichment;
- imported shingle of unsuitable type/size[1] for vegetation development;
- nutrient enrichment through mixing of the shingle and soil layers.

Practical measures to prevent damage or to offset unavoidable damage

There are a number of measures that can be taken, both before the commencement of coastal defence operations and while the work is being carried out. It is important that all contractors are informed in advance of the procedures they must follow to limit any impact on the vegetated shingle.

Preventative measures that can be taken are:

- brief the works unit/contractors before commencement of work to ensure that each individual is clear about the measures being taken;
- require contractor to appoint environmental supervisor to oversee environmental issues (as condition of planning approval);
- limit the number of beach access points for vehicles and if possible, ensure vehicles keep to a limited track;
- clearly define beach access points avoiding any vegetated areas where possible. If the loss of some vegetation is unavoidable, ensure that it is limited to those communities which will be able to re-establish in the relatively short term;
- demarcate any particularly valuable or important area of vegetation prior to commencement of operations to ensure its protection. Use old groyne timber, tape or any other highly visible barrier laid around the perimeter. Keep all vehicles, machinery and people out of the marked area;
- if possible, wash imported marine-dredged shingle to prevent nutrient enrichment;
- if possible, use imported shingle of a suitable type/size[1] for vegetation, at least in those more stable areas where plants are likely to recolonise;
- if possible, limit vehicle movement along the beach to below High Water Mark to prevent impact to vegetation or compaction of shingle on the ridge;
- limit work to the period outside the bird-nesting season. Shingle nesting birds generally nest between March and August;
- include information about the shingle vegetation protection measures that are being taken, in any interpretation on the site.

Trenching on shingle

Experience at Dungeness (Annex 05) has shown that demarcating significant shingle vegetation along the line of any trenching activities with stakes and brightly coloured plastic rope and highlighting the least damaging route ahead of contractors pays great dividends in avoiding disturbance.

Vehicular access

Unprotected vehicular access onto shingle will virtually always cause long-term damage. However, access for water authority or other utilities vehicles will often be necessary. This can be mitigated by the use of temporary trackways across the surface which may well result in bleaching of the vegetation but will markedly reduce geomorphic damage and allow the vegetation to recover in the short to medium term (Dungeness, Annex 05).

Work programming

Work programming is very important. Where possible both sea defence and management works should be carried out at times when disturbance of plants and wildlife are at a minimum. Most vegetation will be least disturbed out of the growing season. Areas of significance for ground-nesting birds such as terns should be avoided even for survey work until reproduction is complete.

[1] Land-sourced shingle tends to have sharp edges and be more angular than marine-sourced shingle, making it less suitable for coastal shingle vegetation. Pebble size is very important in determining which plants can grow. Pebbles with an average size greater than 8cm diameter are normally unsuitable for vegetation. Shingle species generally prefer an average diameter of up to 4cm. Patches of different sized pebbles are the most desirable as they allow a mosaic of vegetated and bare shingle areas to develop. A variety of pebble sizes will also allow a greater number of species to colonise and a series of different matrices to be held in situ.

Coastal defence guidelines and vegetated shingle identification guide.

Many of the above points are contained in the "Good Practice Guide for working on Vegetated Shingle beaches" produced by the West Sussex Vegetated Shingle Project (http://www.pebbledash.org.uk) to help those planning work on shingle areas. In addition, an A5, waterproof, colour identification guide has been produced jointly by the West Sussex Vegetated Shingle Project and East Sussex Coastal Biodiversity Project to enable easy identification of the shingle communities to be protected, by contractors working on site. Similar guidance should be made available for other areas of significant vegetated coastal shingle. These guides have been distributed to all Environment Agency offices nationally with shingle coastlines, and have been widely distributed around the south-east beyond East & West Sussex via the two relevant coastal groups and Standing Conference on Problems Affecting the Coast (SCOPAC).

Emergency works

It is possible to provide guidance, as above, which will help to overcome the damage to shingle habitat during planned operations. However, it is much more difficult to prevent damage when emergency repair work takes place, especially when storms occur and property (and life) is threatened. There are two other points which are important:

Summary guidance - emergency sea defence operations

1. early identification of potential problems, presence of stable mature shingle vegetation, rare plants or animals or breeding birds will help ensure that these interests are taken into account;
2. under emergency circumstances not only is good, accessible information required on the nature of the interest likely to be affected, but this also needs to be available to those undertaking the work. This requires contingency planning including the establishment, in advance, of close working relationships with relevant utility companies, regulatory and other authorities (especially local Environment Agency staff).

Construction Industry Research and Information Association (CIRIA) - the CIRIA guides

The Construction Industry Research and Information Association is an independent, UK-based research association concerned with improving the performance of all involved in construction and the environment. It provides information and advice on best practise and has produced several guides of relevance to this report, including:

- "Environmental Good Practice on Site". This is a very general guide aimed at site managers, site engineers and supervisors (Coventry & Woolveridge 2000) available from CIRIA;
- "Coastal and marine environmental site guide" Research project 646 (Budd et al. 2003).



The coastal and marine guide is specifically designed to inform front-line construction staff working on coastal and marine projects about the environmental impacts associated with a range of common construction operations. The work, funded by (DEFRA and the Environment Agency) provides guidance on avoiding or mitigating these impacts. It is presented in a format suitable for use on site and easy to use in an emergency situation (see above).

The guide is available from the CIRIA Bookshop and can be ordered from their Web Site: http://www.ciria.org/ price £40. CIRIA, 2 Storey's Gate, Westminster, London, SW1P 3AU.

Guidance on environmental impact assessment in relation to marine minerals dredging applications

The potential effects of marine aggregate extraction have been discussed in Chapter 03. A recent review of the Government approach to the control of licensing of marine minerals dredging applications has resulted in greater consideration being given to their potential effect on onshore coastal systems. The following text is taken from MMG 01 and provides guidance on issues that should be considered when assessing the environmental impact of any proposed offshore dredging project. Of particular relevance are Article 6 "Assessment of the Potential Effects of the Dredging Activity" and the requirement to assess the physical impact of aggregate extraction for coastal erosion (through a Coastal Impact Study). In particular they suggest the following should be addressed:

- "*the proposed dredging is far enough offshore for there to be no beach drawdown into the deepened area;*
- *the proposed dredging will interrupt the natural supply of materials to beaches through tides and currents;*
- *the likely effect on bars and banks which provide protection to the coast by absorbing wave energy, and the potential impact on local tidal patterns and currents which could lead to erosion;*
- *likely changes to the height of waves passing over dredged areas and the potential effect on the refraction of waves which could lead to significant changes in the wave pattern*."

This issue is discussed in more detail in Chapter 03. For further guidance on assessing the effects of dredging on the coastline see Report (C505) '*Regional seabed sediment studies and assessment of marine aggregate dredging*' produced by CIRIA (Brampton & Evans 1998). It provides a structured approach to studies of regional seabed sediment mobility, plus guidelines for studies needed to assess the impact on the coastline of proposed dredging operations.

Issues surrounding the use of coastal defence measures to help re-create and restore vegetated shingle are given in Chapter 07.

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Chapter 07 Opportunities for habitat restoration/creation

This chapter sets out to provide some generic guidance on shingle habitat restoration. It builds on the discussion in the previous chapter and identifies a number of States in which shingle can exist. These range from mobile shingle beaches to stable shingle structures and transitions to other habitats. Shingle beaches are precursors to shingle structures, which become stable over time. The beach in its normal state responds to the action of waves and storms, existing in a highly dynamic environment. Once stable the larger shingle structures may remain so for many years, in some cases lying undisturbed for centuries. As has been shown in Chapter 5 human actions have modified or disturbed many shingle beaches and destroyed the surface of large areas of stable shingle.

The first section of this chapter deals with shingle beaches and more stable shingle structures and efforts to re-create more natural, stable forms of the habitat. Where management potentially provides opportunities for restoration of degraded areas of stable shingle, such as in managed realignment schemes, this is also considered. Opportunities for the restoration of shingle areas on the landward side of existing ridges are also discussed in areas where there are transition zones between vegetated shingle and agricultural land. This section of the guide thus deals with the essentially dynamic process of restoration, including areas where extraction of aggregate and/or recycling is taking place. It also addresses the value of natural cycles of erosion by the sea as a means of stimulating restoration.

The second section looks more closely at enhancing biodiversity through artificially creating shingle surfaces suitable for plant and animal colonisation in areas of dry aggregate extraction. Restoration of areas where shallow extraction has resulted in damp conditions is reviewed in relation to the development of semi-natural communities such as those associated with fens. The chapter also highlights the importance of planning ahead before projects i.e. to facilitate seed collection.

Shingle beaches

Restoring shingle beaches and structures is considered for the purposes of this guide separately and in relation to their original stability. Thus two states are identified for **shingle beaches**:

State 1 (erosional) - where a dynamic beach is moving alongshore or landward under the influence of the sea and there is an overall deficit in the sediment supply



Figure 07 - 01 Erosional beach, Porlock, Somerset. Note the complete absence of vegetation on this exposed part of the beach.

State 2 (accretional or semi-stable) where there is no sediment deficit. The beach moves in response to the prevailing wave and tidal regime



Figure 07 - 02 Semi-stable beach, Berridale, Scotland. The beach may accrete and/or retreat but stays more or less in the same location.

Both states are typically associated with shingle beaches and movement between the states forms part of the natural processes associated with this dynamic landform, in which plants can perform a limited role.

The erosional state is often the predominant form and where the shingle represents an important line of sea defence, action to help stabilise the beach is frequently undertaken with management aimed at moving the beach from State 1 - State 2.

The mechanisms involved can include some of the following:

- the construction of sea defence structures;
- beach reprofiling;
- beach nourishment.

The direction of movement is as shown in the Figure opposite. The preferred direction of movement from a sea defence perspective is shown in the figure in blue. Note that in a few cases, and for the restoration of more natural shingle forms, a reversal of the process to restore shingle movement may be considered i.e. State 2 to State 1 by allowing natural processes to dominate. This issue is discussed below and by reference to several cases studies reviewed in the guide: Orfordness (Annex 06); Dungeness (Annex 05) and Porlock (Annex 01).

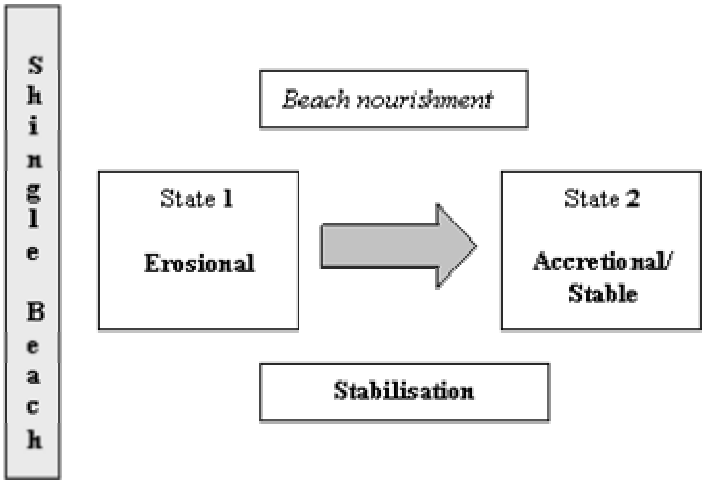


Figure 07 - 03 Principle direction of preferred movement of shingle (from an engineering and vegetation development stand point)

Creating stability - State 1 to State 2, engineering approaches

Undertaking engineering works designed to prevent erosion can, potentially at least, provide the means for creating more stable areas where vegetation becomes established. Both toe protection along the base of the sea walls and breakwaters can be used to reduce scour. Buried revetments, e.g. at Sovereign Harbour in Eastbourne have helped to build up the base of the beach. As a result less material is needed to build the shingle up to a level suitable for flood protection. The top of the ridge is broader, and hence more suitable for the development of vegetation communities. At Pevensey investigations are taking place into the use of tyre bales for a similar purpose. These may have the added advantage that the bales are not solid structures and can provide gaps for the shingle to penetrate and hold the beach together, as well as voids for root systems.

The construction of the Brighton marina in 1972 provided opportunities for a newly deposited shingle beach to form between a groyne and the western breakwater. Detailed studies of this area (Black Rock Beach) showed, for example, how vegetation can develop once engineering activities cease (Packham & Spiers 2001). Between 1975 and 1996 the beach remained relatively stable and both typical and rare plants colonised, helping to create an unusual community which included the sea knotgrass, *Polygonum maritimum* (a plant of the Mediterranean, rare in Britain and at or near its northern range here). This community persisted until a storm in 1996 washed it away.

The study at Black Rock provides some pointers to future approaches where construction is used to protect a beach. If left undisturbed once construction is complete, then it is possible for important and

rare plant communities to develop. Increasing the period of beach stability allows the development of a type of matrix and hydrology suitable for the establishment and survival of the vegetation.

The case of Black Rock is unusual, in that access to the beach by machinery was limited. More usually sea defence structures are associated with nourishment and/or reprofiling and such development is not possible. This leads to consideration of the implications of artificially engineered approaches to beach protection and their impact on coastal vegetated shingle. The implications of these approaches for the maintenance or restoration of coastal vegetated shingle are discussed next.

Creating beach 'stability' with beach reprofiling

Generally, reprofiling beaches destroys any vegetation in the areas where work takes place. The nature of the beach slope is important, as is the height above sea level. In sea defences conventional approaches involve raising the crest height, which at the same time tends to steepen the foreshore. This has happened at several sites where loss of vegetated shingle has occurred, e.g. Porlock (Annex 01) Cley-Salthouse (Annex 07) and the sites on the south coast (Annex 03, 04 and 05). The evidence suggests that this approach is not wholly sustainable from a sea defence point of view. This is particularly illustrated by the work at Porlock (Annex 01) and Cley-Salthouse (Annex 07).

Comment: *A major question arises about the use of coastal defence structures and beach reprofiling/nourishment as to their appropriateness within important nature conservation sites. Sites with a dynamic balance between stability and instability allows for a variety of communities to develop, including those associated with both ephemeral and more stable structures (Chapter 02). Thus creating stability must be considered in relation to compliance with the 'Habitats' Directive and in relation to the Habitat Regulations (dealt with in Chapter 04). The designated sites have fixed boundaries, with lines on maps used to define the protected areas. This can impose a rigidity on restoration activities, which is not commensurate with the inherent dynamism of the systems. This is particularly important where shingle beaches or structures move in response to tidal energy and storms, beyond the boundary of the designated site. These issues are discussed in more detail in the general guidance in relation to the coastal vegetated shingle Habitat Action Plan below.*

Beach profile

The creation of a natural beach profile is crucial to the establishment of shingle vegetation. A study of regeneration of vegetation (Walmsley & Davy 2001) suggested that recontoured beaches should be left to allow natural reshaping by winter storms before any attempt to plant vegetation is made.

As a guide, the slope and sediment size best for restoring vegetation on shingle beaches, based on a natural beach profile, has the following characteristics:

Feature	Size and range	Comment
Slope	1:100 - 1:15	Natural range of slope
Sediment size	2mm - 200mm	Range for shingle used to define habitat (Randall 1977)
Sand content	10 - 20% sand content	Best for vegetation establishment (Walmsley & Davy 2001)

Beach nourishment

Beach nourishment with shingle is frequently used to improve its sea defence capability. There are numerous examples, especially on the south coast of England where the Environment Agency and others are undertaking such work. If a suitable source of material exists nearby, beach nourishment with shingle is a relatively easy process. It can also, if the new or replenished beach is allowed to develop without interference through reprofiling and the material is washed to reduce nutrient enrichment, provide the foundation for restoration of vegetation. In many instances this is not the case and continued bulldozing of beach material to increase crest height and 'improve' the sea defence capability prevents such colonisation.

Successful and long term effectiveness of beach nourishment without environmental damage and with opportunities for colonisation can occur. These will depend on the nourishment process taking into account the following:

- the nature and type of beach appropriate to the location (slope, width, sediment type and sediment size, see Chapter 06);
- the role of nearshore tidal currents in moving shingle across the shore and in a longshore direction i.e. the beach response;
- the long term wave climate and extreme storm sea-states, which combine large waves and high water levels;
- the relative rate of sea level change.

Under most circumstances the value of the beach to the protection of life and property is the primary reason for nourishment. Thus most schemes depend on an assessment of the following:

- risks (human, economic, environmental), both locally and along adjacent frontages;
- costs of the scheme, including initial placement and long term maintenance; the availability of appropriate material (dredged aggregate, quarried gravel, crushed rock) in the volumes required (both short and long term) and the influence of different recharge sediment gradings on the cross-shore and long-shore processes;
- potential impacts of the scheme on the human and natural environment (e.g. recreation, archaeology, nesting sites, fisheries, outfalls).

Assessment of the appropriateness of such schemes also depends on the suitability of the material used (grain size, freedom from pollutants etc.). Where material is derived from marine sources they are also subject to a Government View procedure, which controls the extraction of aggregates from offshore. This procedure requires an assessment of the potential impact of extraction on nearshore and coastal processes (dealt with in more detail in Chapter 04).

Problems potentially arise from a conservation perspective with all of these solutions to preventing erosion. This is particularly acute in areas where more stable beach forms exist, which have both ephemeral strandline plant communities (with associated and sometimes rare species of both plants and invertebrates) and more stable structures and their communities. Thus although engineering approaches to stabilising shingle areas can help to create or re-create vegetated shingle habitat, it rarely does so! Recently other approaches are being investigated, e.g. the use of buried tyre bales at Pevensey as noted above.

However the socio-economic imperative for the protection of life, land and property remains a dominant one. Opportunities for using the natural predilection for shingle beaches to respond to tides and waves including storms and by so doing provide opportunities for habitat re-creation have only recently been considered at all, except in a few circumstances. This issue is considered in more detail in the section on the implementation of the Habitat Action Plan aims below.

Summary guidance - creating stability, engineering approaches

It is clear that beaches can be stabilised using artificial structures. Because these are associated with coastal protection measures, other activities usually take place, such as beach replenishment and/or reprofiling. As a result the establishment of new shingle communities is very rare and the case of Black Rock above may be an isolated example.

Restoration of stable shingle and vegetation

The natural process of beach ridge formation (Chapter 02) can result in the development of large structures as former beaches are piled against each other. These structures may take a variety of forms depending on strength and direction of the waves, storm conditions and availability of sediment in a suitable size range. Shingle structures may enclose estuaries and embayments as a result of material being moved by long shore drift and form spits, bars and barriers. Major accumulations of successive ridges piled on shore under storm conditions can develop into cuspate forelands. Nesses and recurved spits form where there is a convergence (or divergence) of littoral drift. Cheniers are a special form of shingle which is driven onshore as a series of low shell deposits (Chapter 02).

In the absence of human interference these all provide essentially stable surfaces on which other factors come into play, which influence the development of vegetation. The action of the waves and storms that create the ridges help sort the pebbles (Chapter 02). Subsequently material is washed into the spaces between the pebbles and can result in sufficient water retention to facilitate plant growth. Over long periods of time stable communities develop, which can range from lichens growing directly on the pebbles (as at Orfordness), to grassland, heath and scrub (State 3, Figure 07-04 below).

State 3 - (Stable). Shingle ridges removed from the influence of storms and where the build up of material between the pebbles helps support plant establishment



Figure 07 - 04 Stable shingle ridge and hollows, Culbin Shingle Bar, Highland Region, Scotland. Note the build up of vegetation on the sides and hollows of the ridge, where the spaces between the shingle become filled with organic and other material and improve moisture retention opportunities.

This state provides geographically the widest and most diverse expression of the vegetated shingle habitat in Great Britain, as described in Chapter 02.

As we have seen in Chapter 05 human action can seriously damage or disrupt this surface vegetation. Typically large expanses of shingle provide a ready source of accessible gravel for a variety of building purposes. Excavation of the shingle destroys the surface vegetation and any associated interests. The extent and nature of any restoration depends on whether the excavation is above or below the water table. Where disturbance or excavation remains above the water table for most of the time (State 4, see below) then vegetation can, potentially at least, be restored on the shingle surface, although stable communities are likely to take a considerable time to revert to their natural state.

State 4 (Disturbed/Excavated) A state where the depth of the disturbance / excavation is above the water table



Figure 07-05 Disturbed/excavated shingle at Kingston, Moray Firth, Scotland dominated by herbs including *Shoenus nigricans* in the foreground, a species of damp places.

NB. A distinction is made here between disturbed shingle surfaces which remain dry and those where the surface is close enough to the water table to provide opportunities for good plant establishment and growth. In such instances such as at Kingston (Spey Bay Shingle Annex 08) and at Dungeness (Annex 05) species-rich secondary communities can develop.

Restoring these stable surfaces essentially revolves around attempts to re-establish vegetation on stable shingle surfaces. Restoring the physical shingle structure (ridges and lows) is a precursor to re-establishing vegetation, but is difficult, especially on dry well drained shingle. Figure 07-06 gives a summary view of the direction of the main restoration opportunities. Several approaches for restoring vegetated shingle (moving from State 4 to State 3) are considered next.

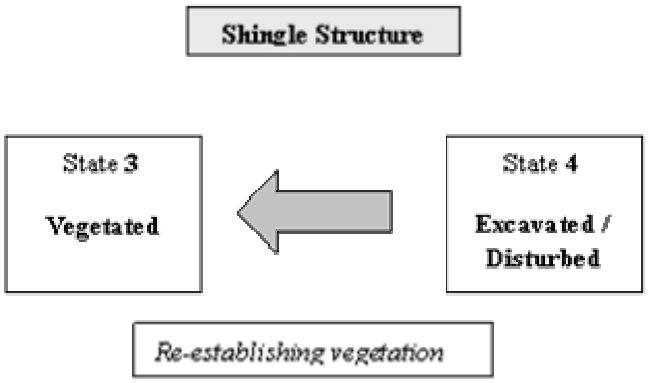


Figure 07 - 06 Principle direction of preferred movement of shingle (from a conservation perspective, which aims to restore vegetated shingle habitat)

Re-grading disturbed shingle

As a first stage in restoring vegetated shingle, re-creation of a suitable shingle matrix is needed. Few attempts have been made, as far as can be ascertained, to artificially re-create the surface shingle form of ridges and hollows. One project carried out at Orfordness involved the grading of shingle to mimic the process undertaken by the sea (Annex 06).

The damage to the shingle at Orfordness has been considerable. Although excavation has been relatively restricted, loss and damage of the surface has been extensive at the northern (ness) end of the structure. This has destroyed the vegetation and severely degraded the morphology, such that natural regeneration of the vegetation is very slow or non-existent. [A requirement for the development of vegetation on the extremely porous shingle surface lies in the presence of fine fractions between the pebbles, which allow the retention of moisture and facilitates plant seed germination. This fraction is lost with the surface disturbance of the shingle.]

The National Trust undertook to test, experimentally, whether it was possible to regenerate shingle flora on some of the worst degraded and damaged sites. The project was carried out in 2000 as part of the European Union LIFE-Nature project 'WILD NESS - The Conservation of Orfordness, Phase 2'.

The chosen site was selected in an area substantially degraded by military use. Initial work on the ridge to be restored involved the scraping off of the surface shingle to a depth of approximately 18-20cms (below the depth to which germinating seeds will reach).

"The shingle was graded, using an adapted small commercial screener, into four nominal sizes <5mm, 5-15mm, 15-25mm and >25mm. These measurements are representative of the natural sizing recorded on adjacent ridges. This material was then manually and mechanically replaced onto the scraped area in order to reproduce the height, width, spread and size ratios of a section of undamaged ridge adjoining the test area." Five treatments were tried; see Table 07- 01 below:

Section 1	Restored sizing and grading plus added fines and added *Silene maritima*/*Arrhenatherum elatius* seed
Section 2	Restored sizing and grading plus added fines
Section 3	Restored sizing and grading plus added *Silene maritima*/*Arrhenatherum elatius* seed
Section 4	Restored sizing and grading only
Section 5	Existing 'undamaged' ridge - possible source of colonisation

Table 07 01. Five experimental treatments to restore shingle vegetation.

Recolonisation

The work at Orfordness is being monitored using presence or absence linear transects and fixed point photography. There was no evidence of recolonisation by October 2002 when the photograph opposite was taken (see Figure 07-07), some 2 years after the work was undertaken.

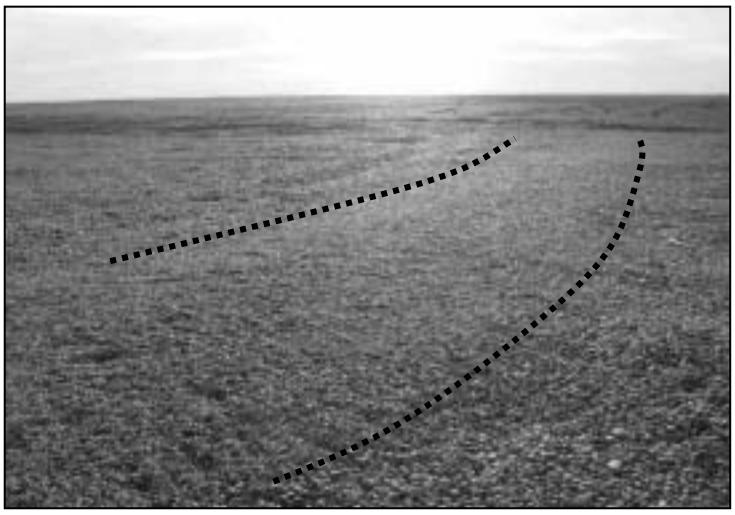


Figure 07 - 07. Site of experimental shingle restoration. The location of the restored ridge is indicated in the picture.

Part of the LIFE Nature funded project: "**Wild Ness**: the conservation of Orfordness, Phase 2 (LIFE97 NAT/UK/004245)".

Summary guidance - re-creating shingle ridge profiles

The evidence so far suggests that regrading shingle is feasible but costly in time. The re-establishment of vegetated shingle does not seem to readily occur even with the introduction of seed (note this comment relates only to this experiment, but see also below, "Restoring Vegetation"). From this work it appears the natural forces of tides and waves are much more efficient tools for sorting coastal shingle than anything human restoration can achieve.

Restoring vegetation - free-draining shingle surfaces

Research has also been applied to recreating shingle vegetation without the grading of the shingle substrate. Restoring vegetated shingle surfaces is principally a nature conservation issue. Unlike sand dunes (and to some extent saltmarsh) plants play a limited role in stabilising the structure itself. There is little information available on how to re-establish plants on shingle, though a recent series of studies at Sizewell in Suffolk has elucidated several approaches, namely:

- natural regeneration;
- use of the natural seed bank;
- use of sown seed for restoration (Walmsley & Davy 1997);
- use of container grown plants (Walmsley & Davy 2001).

[*These methods are discussed in more detail in the "Good Practice Guide for habitat restoration, re-creation and creation on the coast". Note small-scale projects are being trialled in East & West Sussex; see also reports for Pagham, Annex 03 and Rye Harbour, Annex 04.*]

Much is still to be learned about vegetation restoration on shingle. Walmsley and Davy (2001) in reviewing the work of the 'Sizewell B' Vegetation Restoration Project where mature, container-grown plants were able to survive best because they possessed extensive root or rhizome systems. Trials are being undertaken on several sites on the south coast. On a small area at Bognor Regis planting was carried out successfully in 2001/2002 and seeding in 2002 (results are not yet available). Following extensive coastal defence work in the early 1990s, seeding was carried out at Elmer, Middleton-on-Sea. Although not scientifically monitored, the beach rapidly developed a vegetated shingle community. However, it is uncertain exactly how much was due to the seeding application and how much to natural re-colonisation. The site is now being regularly monitored but unfortunately too late to observe the early stages of colonisation. Other trials are proposed at Selsey Bill near Pagham with a variety of seeds and at Dungeness with *Cytisus scoparius* but the outcomes of these studies will not be available for 2-3 years (Annexes 03 and 05).

Whether or not seed or other plant material is introduced the structure of the shingle surface is all important. The creation of a natural beach profile is crucial to the establishment of shingle vegetation. Recontoured beaches should be left to allow natural reshaping by winter storms before any attempt to plant vegetation is made (Walmsley & Davy 2001). In this context the development of soils with water retention characteristics cannot be over emphasised (Chapter 02, soils).

Overall this approach can help with the re-creation of vegetated shingle though the success is relatively limited when compared with the cost of undertaking the restoration. At the same time storms can (as might be expected) undo the work in a short space of time, on the most seaward sections of the restored beaches.

Summary guidance - re-creating vegetation on dry shingle surfaces

Natural regeneration of plant communities takes place over time and a variety of techniques have been tried to encourage the establishment of vegetation on dry degraded/disturbed surface shingle. These do not appear to have been particularly effective, to date. Hence restoring vegetation to stable shingle (moving from State 4 to State 3) may largely be a matter of 'leaving nature to take its course', see below.

Restoring shingle vegetation - waterlogged surfaces

Although there are no specific examples of active intervention to create or restore vegetation on shingle surfaces which have been excavated to levels at or near the water table, there are a few sites where such conditions exist. At Kingston (Spey Bay, Annex 08) on the south shore of the Moray Firth excavation has created depressions in the surface shingle that have become colonised by a variety of plants typically associated with damp hollows. These communities may be particularly rich in species including the rare coralroot orchid *Corallorhiza trifida*. At Dungeness (Annex 05) shingle spoil and areas of shallow excavations have also been colonised by a variety of communities more typical of damp conditions, including some similar to those of wet sand dune slacks. Some highly disturbed shingle; with an input of fine-grained material along road verges, can have a much richer complement of plant species than natural undisturbed grassland on shingle (Ferry 2001).

Summary guidance - re-creating vegetation in damp hollows on shingle

Restoring vegetation on shingle surfaces at or near the water table is more readily achievable, based on the limited examples from sites in the UK. The presence of moisture (a limiting factor in most undisturbed stable shingle) and the apparently more rapid build up of humus help to create suitable conditions for plant establishment and growth. Although there are no known examples, an approach which seeks to lower the level of already disturbed shingle at the margins of these areas could help create transitional communities such as those found on the shingle at Kingston, Scotland (Annex 08) and Rye (Annex 04) where they resemble dune slack communities. The disturbance associated with mixing shingle with fine-grained material as has occurred at Dungeness provides another potential method for habitat enhancement see Ferry (2001).

These examples suggest that it may be possible to create or re-create these communities which are reminiscent of transitional zones between open dry shingle and other habitats sustained by springs or with soils with water retaining properties.

The role of gravel pits

A fifth state has been included in this guide (rolled into State 4 above in the "Good Practice Guide for habitat restoration, re-creation and creation on the coast") which takes account of those areas where excavation is at or below the water table, for all or most of the time.

State 5 (Excavated) Below the water table helping to create coastal gravel pits (Figure 07-08)



Figure 07 - 08 Newly excavated gravel pits at Dungeness, Kent on land owned by the Royal Society for the Protection of Birds (RSPB). A large part of these areas have subsequently become part of an extensive wetland nature reserve. The picture was taken in 1980 before the RSPB stopped selling gravel-winning concessions at the site.

As we have seen in Chapter 05 pressures on the excavation of onshore deposits of shingle are enormous. This is the most usually encountered and often most extensive form of habitat loss. In these cases the shingle void, because it is excavated to below the water table, results in open water 'gravel

pits'. Generally, excavated shingle areas are considered to have a reduced value for nature conservation because of the loss of the specialist and rare plant communities that make up the vegetated shingle habitat. There are, however, replacement values associated with the open water and the creation of new wetland habitat such as the creation of roosting, feeding and breeding habitats for birds, and the creation of saline lagoons. Restoration of the gravel surface and vegetated shingle (including those communities defined in the 'Habitats' Directive) is not possible. The replacement values associated with developing this habitat are thus not comparable with the restoration envisaged in moving from State 4 (Disturbed/Excavated) to State 3 (Stable).

Information on restoring gravel pits is available, see particularly the Royal Society for the Protection of Birds' publication "Gravel Pit Restoration for Wildlife: a Practical Manual" 1990 (ISBN 0 903138 603, RSPB Code 24-015, Price £12) and the "Gravel Pit Restoration for Wildlife: Site Managers Guide" 1990 (RSPB Code 24-014, Price £5).

Summary guidance - gravel pits

The replacement of stable and undisturbed vegetated shingle ridges with excavations, which result in areas of open water, is almost always a retrograde step. The new habitat and coastal bird populations that can colonise the areas do not represent an acceptable trade off for the loss of the rare coastal vegetated shingle habitat.

General guidance and the UK vegetated shingle Habitat Action Plan (HAP)

Naturally, shingle coasts are dynamic and tend to migrate along the coast or landward. They have at the same time become a major source of material for the construction industry, sites for building infrastructure such as nuclear power stations and houses and as a recreational resource. They also have important 'natural' sea defence capabilities. Even if this capability is recognised they are often augmented by artificial structures, in the belief that this is essential for the protection of life, land and property.

Vegetated shingle beaches and structures are important habitats within the European Union 'Habitats' and 'Species' Directives. The values associated with this habitat and described in Chapters 01 and 02 have, not only been damaged and destroyed by human activities (Chapter 05) but have also come under pressure as a result of changing environmental conditions, driven by global warming and sea level rise.

Vegetated shingle is listed as a priority habitat under the UK Biodiversity Action Plan. The Vegetated Shingle Habitat Action Plan recognises two principle issues (2.1 and 2.2 below) relating to sea defence and the sustainability of shingle habitats:

"**2.1 Sediment supply**. *The health and ongoing development of a shingle feature depend on a continuing supply of shingle. This may occur sporadically as a response to storm events rather than continuously. It is frequently lacking owing to interruption of coastal processes by coast defence structures, by offshore aggregate extraction or by artificial redistribution of material within the site (e.g. Dungeness). Attempts have been made to rectify the situation by mechanical reprofiling, which is likely to fail in the long run because it does not address the lack of new material, or by beach recharge*."

Added to these considerations must be the fact that most shingle structures and many shingle beaches are derived from relict sources, i.e. relative to the quantities present today there is little or no new material (Chapter 02).

Comment: *This study has clearly identified the importance of understanding the **relict nature** of many of the shingle deposits that are present today. The conclusions from Futurecoast summarised in the report on Slapton (Annex 02), highlight the continuing need for beach nourishment on the south coast and the deficit of material identified for Dungeness (Annex 05 and 05a) tend to confirm the view expressed in 2.1 above that in **many instances engineering solutions to shingle erosion are unsustainable in the medium to long term**.*

Erosion as a healing force in degraded areas

This leaves the consideration of approaches which 'work with nature' and accept the 'naturally' dynamic state of shingle beaches and many of the shingle structures as providing the potential for a more cost effective approach. This recognises the second key principle articulated in the vegetated shingle Habitat Action Plan, namely:

"**2.2 Natural mobility**. *Shingle features are rarely stable in the long term. Many structures exhibit continuous longshore drift and ridges lying parallel to the shoreline tend to be rolled over towards the land by wave action in storm events. This movement has a knock-on effect on low-lying habitats behind the shingle. Movement is likely to be accelerated by climate change resulting in sea level rise and increased storminess.*"

Taking these two principles into account a question arises as to the acceptability of allowing 'nature to take its course' when the re-creation of one habitat will be at the expense of another. Section 5.3.3 of the HAP expresses the dilemma as follows:

"*Allowing natural landward movement of shingle features* will, in some cases, affect other habitats such as saline lagoons, grazing marsh, fens and reedbeds, some of which will be designated sites. The implementation groups for the relevant HAPs should be advised on how to make appropriate provision for habitat creation. In some cases, breaches in shingle banks may lead to the development of saltmarsh habitats and this needs to be taken account of in the respective HAPs.*" [* Advocated in 5.1.1 of the HAP]

Letting nature take its course

The situation at Porlock (Annex 01) is perhaps the best illustration of the way things can develop under more or less natural conditions. Here a natural storm breach in effect pre-empted discussions about continuing to protect the shingle ridge through reprofiling and beach nourishment. Today the former tidal land converted to grazing marsh and arable land has now reverted to brackish and saltwater communities as the tide has reinvaded the land following a breach in the shingle ridge caused by a storm. The site was re-designated as a Site of Special Scientific Interest following the breach, with a new description reflecting the fact that saltmarsh and transitions to brackish water habitats had largely replaced the coastal grazing marsh. The site shows that coastal habitats can be re-created and that although they may be different from that which they replace, they may be no less important.

Other considerations are relevant at this site, not least the attitude of the local population, opposed to leaving things as they are i.e. NOT closing the breach and reinstating the sea defence structures. The site report (Annex 01) discusses this issue also.

Summary guidance - the example of Porlock (Annex 01)

This case clearly illustrates a possible way forward when existing protection measures are thought to be unsustainable. The lessons learned from here should be communicated and applied more widely. This is not only because of the nature conservation implications but also those associated with social attitudes to 'abandoning' the land to the sea.

The shingle squeeze

A further issue arises where 'natural' processes result in habitat loss, for example through 'coastal squeeze' described both for Slapton (Annex 02) and Cley (Annex 07). Where these form part of a candidate Special Areas of Conservation or Special Protection Areas is it appropriate to 'let nature take its course'? With no interference it is conceivable that the Favourable Conservation Status of the selected habitats and species would be compromised and the overall biodiversity of the site diminished. This would not only compromise the overall value of the area but could also work against achieving biodiversity targets set for example within a 'Natural Area' as defined by English Nature.

In the UK the Conservation (Natural Habitats, & c.) Regulations 1994, as amended, implement the European Union 'Habitats' and 'Birds' Directives. In particular, "The Habitats Regulations" transpose into UK law the provisions of the European Union Habitats Directive and introduce stringent requirements for the conservation of Special Protection Areas and Special Areas of Conservation (including candidate SACs). [For more detail on the regulations see Annex 01.] Application of Regulations 48 and 49 of the regulations if strictly interpreted create a strong presumption that habitats be preserved in their present location. Only where "no alternatives" and "imperative reasons of over-riding public interest" are demonstrated is providing compensatory habitat an acceptable alternative.

At Cley the original realignment scheme (Annex 07) envisaged protecting some of the existing interest (mainly reedbeds for their bird interests) at the same time as allowing the 'natural' restoration of the shingle ridge following the cessation of beach reprofiling. The reconsideration of the scheme has led to the recognition that although leaving the system to reassert itself with minimal interference will lead to the loss of some habitats, others will develop. Overall it seems likely that the conservation value will be maintained and the site as a whole will remain in 'favourable condition'. A similar question has arisen at Slapton Sands (Annex 02) where the continued roll-over of the shingle beach will ultimately encroach on a coastal freshwater lagoon reducing its area and changing its characteristics.

Summary guidance - 'shingle squeeze' and the 'Habitats' Directive

From these examples it seems entirely appropriate to consider that the loss of one habitat at the expense of another is entirely in keeping with the desire to promote more 'natural' and dynamic ecosystems. The dilemma as expressed in 5.3.3 of the coastal vegetated shingle Habitat Action Plan, quoted above, identifies the need to consider the re-creation of replacement habitats elsewhere. Where these are different from the vegetated shingle and include other coastal habitats reference should be made to the "Good Practice Guide for habitat restoration, re-creation and creation on the coast".

Initiating erosion for conservation purposes

The reliance on a 'static' approach to conservation, which is so often associated with the identification of boundaries of sites on maps, may be counter productive for a habitat existing in such a dynamic environment. As expressed above the natural tendency for movement both alongshore and inland not only has implications for the management and restoration of beaches and areas of vegetated shingle, but also associated habitats such as lagoons and saltmarshes. Whilst natural movements of shingle in response to storms will be seen as being part of the "natural structure and function" of the system, interference to create instability may not!

These issues have particular poignancy in relation to Dungeness and Orfordness. Here the scale of damage to the surface shingle in some parts of the site covers such a large area that any of the approaches to restoration mentioned above would be likely to be ineffective. Initiating erosion or at least removing those static artificial features such as groynes and breakwaters and/or ceasing beach nourishment could help bring about major restoration of shingle beaches and ridges. This would, of course, require very careful consideration of the implications for the land and property formerly 'protected' from erosion and flooding by the coastal defence activities. In the case of Dungeness

(Annex 05), for example, this may be prevented because of the need to protect the nuclear power stations, but perhaps not at Orfordness (Chapter 06 and Annex 06)?

Summary guidance - creating instability

The past reliance on coastal protection measures to prevent shingle beaches and structures eroding has helped to create and sustain a much impoverished habitat. In some instances the area of damage is so large that even if the restoration techniques described above were to work and restore form and function to the shingle and with it vegetation and associated animal life, the resources needed would be considerable. Under these circumstances and on the larger sites, creating instability, or at least letting natural forces reassert themselves is probably the only option.

Comment: *Taking the above course of action will not be easy. As a general rule, however, wherever possible from a nature conservation perspective,* ***shingle structures should be left entirely alone****. In the majority of circumstances, geomorphological change promotes environmental diversity (Randall & Doody 1995). Given the historical reliance on engineering approaches to protection this approach* ***will require considerable persuasion****, not least amongst the local population. The situation at Porlock (Annex 01) and the potential examples of Cley (Annex 07) and Orfordness (Annex 06) may provide important pointers for this changing paradigm.*

The issues discussed in this section, which relate to allowing or promoting change for conservation purposes, especially where the loss of one habitat may be at the expense of another, are being examined in some detail in a current study "Options for maintaining features of European Importance in dynamic coastal situations" being undertaken as part of English Nature's Living with the Sea project (See: http://www.english-nature.org.uk/livingwiththesea/). For this guide the options are considered in more detail in relation to Orfordness (Annex 06) and the Cley-Salthouse realignment scheme (Annex 07).

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Chapter 08 Monitoring

There are three key stages in any process which aims to monitor the status of any habitat as follows:

1. **survey** to establish the location, scale and value of a particular resource;
2. **surveillance** to identify unforeseen change and where appropriate ensure compliance with agreed legislative or other control mechanisms - simply 'watching to see what happens';
3. **monitoring** to help identify and assess the effect of management policy or action on a given set of parameters and give feedback on their effectiveness.

The purposes of survey and monitoring

Survey is used to assess the status of shingle and shingle vegetation. This is affected through habitat surveys and the like. Whilst strictly speaking it is not monitoring as such it is an essential prerequisite for any future attempts to assess change and the influence of policy and management action on this habitat. In particular surveys attempt to:

1. establish the location and extent of coastal vegetated shingle;
2. identify the condition of individual sites (included in this are Sites of Special Scientific Interest and candidate Special Areas of Conservation). Because nearly all shingle vegetation is included within Annex 1 habitats of the EU 'Habitats' Directive, base-line monitoring is required to confirm its status. This will add to the data already available and will be the precursor to monitoring for other purposes.

In its turn monitoring is necessary to:

1. assess current condition of Sites of Special Scientific Interest;
2. to record the success of habitat restoration or creation;
3. to examine the on-going effects of coastal management works on coastal vegetated shingle;
4. to measure the impact of flood management schemes on shingle habitats, particularly where creation/enhancement works have been carried out.

Survey techniques

Geomorphic survey and monitoring

At some sites it will be important to measure shore mobility. This is best carried out using a series of fixed transects orthogonal to the shoreline. These can be resurveyed at intervals using a dumpy level or theodolite. Ideally a useful body of data can be amassed by surveying monthly but even an annual monitoring of change is helpful for defining management objectives.

Neal *et al.* (2001) highlighted the value of ground-penetrating radar in the geomorphic study of coastal shingle structures. These techniques avoid the cost and damage of excavation and increase the historic knowledge of shingle structures, which is so significant in prediction.

Hydrology and soils

Monitoring the hydrology and soils of coastal shingle is complicated both by the large interstitial spaces between clasts and the difficulty of digging standard soil pits in such sediments. These problems are addressed by Burnham & Cook (2001) at Dungeness and for shingle soils more generally by Sneddon (1992). Burnham and Cook discuss the methods and values of borehole data and review the use of lysimeters in hydrological monitoring.

There is a lack of research on the edaphology of shingle substrates. Ferry et al. (1989) measured pH, sodium, potassium, calcium, magnesium and phosphorus with respect to vegetation type and along a

transect from maritime to terrestrial shingle at Dungeness. The patterns offer evidence of significant leaching. Fine sediment, pH and nutrient ion concentration variation were important in explaining variation in vegetation.

Soil survey and monitoring is difficult on shingle because of the thin soil (commonly 0-10 cm but 30 cm maximum) which precludes standard collection methodologies within different horizons. This may be overcome by collecting material from the full depth of the limited profile. The inert nature of larger sediments makes analyses best conducted on interstitial material only (under 2 mm). Results must therefore be expressed not as a unit volume but with reference to percentage weight of fine fraction (Randall 1977).

The most significant factors to monitor in shingle substrates are silt/clay fraction as this gives an indication of soil-moisture retention; pH, which is a surrogate nutrient absorption; chlorides because of the proximity of the sea; macronutrient exchangeable cations, potassium, calcium and magnesium; and sodium which, due to salt spray, can be an important factor in shingle environments. Organic matter should also be monitored due to its wider implications for soil texture and its importance for cation exchange. Techniques of analysis are described by Brady (1974) and Courtney & Trudgill (1984). Shingle soils have been found to be highly variable and unique (Sneddon 1992).The shallow soils offer only a small reservoir of nutrients for plant growth and successful nutrient cycling is therefore vital. Management practises must take careful consideration of activities which may disturb the soil as there is a great potential for completely destroying thin soils of this nature.

Coastal vegetation and the National Vegetation Classification

Rodwell (2000) describes the most comprehensive coverage of the present knowledge of coastal vegetation in Britain. The techniques used and approach to data analyses are given in an "Introduction" to each volume of British Plant Communities. Stands of vegetation were chosen on the basis of their relative homogeneity in composition and structure. Data were recorded in quadrats of 2 x 2 m for short herbaceous vegetation, 4 x 4 m for taller or more open vegetation and 4 x 2 m for linear vegetation. Samples were recorded on standard sheets and subjected to standard analytical procedures. They are presented as phytosociological tables of community types on roughly the same scale as a 'Braun-Blanquet association'.

Sampling and analysis of shingle vegetation

The most relevant section of Rodwell (2000) for shingle is included within the section on 'shingle, strandline and sand-dune communities'. However, Rodwell admits that "*coverage of shingle features around the coast (is) less (than) adequate and ….we did not incorporate developing surveys of Dungeness by Dr Brian Ferry and of very many shingle beaches around the coasts of England and Wales by Pippa Sneddon working with Dr Roland Randall. Probably at least one further community could be added to our account from these surveys*."

Strandline and shingle communities

Rodwell (2000) recognises two assemblages, dominated by ephemeral, nitrophilous herbs which make a brief fragmentary appearance during the growing season on shingle where organic debris has been dumped along the strandline. On warmer, southern coasts the *Honckenya peploides-Cakile maritima* community (SD2) is present. Towards northern Britain the *Matricaria (Tripleurospermum) maritima-Galium aparine* community (SD3) replaces this. These vegetation types are regarded as pioneer communities that are overwhelmed by storm conditions and re-establish periodically as conditions become suitable. These are some of the 'annual vegetation of drift lines' legislated under Annex 1.

The only other assemblage included by Rodwell (2000) from coastal shingle is the *Rumex crispus-Glaucium flavum* community (SD1). This is essentially a southern British community. It is an example of the 'perennial vegetation of stony banks' also legislated for under Annex 1 of the EU 'Habitats' Directive.

In the UK the Annex I Annual vegetation of drift lines (1210) type is not always easy to classify using the National Vegetation Classification. It consists, as outlined above, of driftline vegetation on stony

substrates and other coarse clastic sediments and can include the SD2 and SD3 NVC types. MC6 *Atriplex prostrata – Beta vulgaris* ssp. *maritima* sea-bird cliff community and other vegetation types not described in the NVC, e.g. monospecific stands of *Atriplex* spp. may also occur. Drift-lines on sandy shores are assessed as sand dune communities (see 2110 Embryonic shifting dunes), and are not included in this Annex I type. However, where drift line vegetation develops on other coarse clastic sediments, such as shell-banks (cheniers, Chapter 03), it can be considered as part of the Annual vegetation of drift lines.

JNCC's coastal survey programme

Although the NVC description of British Plant Communities is valuable at a national scale for all vegetation types, it was felt that a more detailed survey of British shingle was necessary to understand the variety of vegetation present. A survey of Dungeness was carried out by Ferry et al. (1990) and a survey of the other major shingle structures in Great Britain was completed by Sneddon & Randall (1993a, b, 1994a, b).

Methodology

Sneddon & Randall identified around sixty shingle structures and beaches supporting a permanent flora above the strandline. Each was identified using habitat maps; information supplied by English Nature Regional staff and published sources (Figure Chapter 08-01). Each site was surveyed using the framework of the National Vegetation Classification described above, and field techniques were based on those recommended for that survey.

Sites were firstly surveyed by eye to identify stands of homogenous vegetation to be used as mappable units. Within these stands, vegetation was sampled using a 4 x 2 m quadrat, found to be the most appropriate size for the vegetation types encountered and consistent with the quadrat size previously adopted by Ferry & Walters (1985) at Dungeness. Wherever possible a minimum of five quadrats was placed in each stand of vegetation. However, in some cases size constraints permitted only a smaller number.

All species of vascular plants, bryophytes and lichens (excluding saxicolous lichens) were recorded for each quadrat and each species was allocated a cover/abundance score using the Domin scale. In addition soil depth and pH were noted along with vegetation height and any evidence of grazing. Target notes were used to describe any smaller features of interest, either physical or biological, which may provide a useful supplement to the quadrat data.

Site data, such as land use and any forms of disturbance, were collected at each site, while additional site information such as % of SSSI cover and past land use were recorded, based on information collected prior to fieldwork.

The quadrat data were entered into a computer programme that organised them into classificatory units suitable for mapping. The programmes used were TWINSPAN and TWINTAB as specified by NVC. These packages combine quadrats of similar floristic composition into groups and these groupings were then compared with those already identified by NVC keys and tables. These units were then used for mapping. The original data analysis was conducted on a data set of 3250 quadrats. Two methods for mapping were employed in the field. Where vertical aerial photographs were available at a suitable scale they were used directly: elsewhere units were sketched onto enlarged 1:10000 maps of the sites on which the position of individual quadrats were located.

Results

Twinspan analysis is a particularly valuable technique for classificatory purposes because it uses reciprocal averaging to define clearly axes of dissimilarity within data sets (Causton 1988). The original analysis identified 170 quadrats that readily subdivided further but left 3080 quadrats somewhat difficult to subdivide. The 170 quadrats were therefore excluded as they were readily identified as similar to NVC saltmarsh or maritime cliff communities. Such communities are found on shingle where it is adjacent to marsh sediments or where the shingle contains relatively high levels of sea bird guano.

A second run of analysis on the 3080 quadrats gave a stable first division of 948 - 2132 quadrat separation. Indeed, this analysis continued to be robust with few classificatory end groups emerging before the sixth level of division and with most end groups containing less than 30 quadrats by the tenth level. Where over 50 quadrats still remained together further analyses were conducted to test the level of homogeneity and in some cases a further division of classificatory units was accepted.

The mapped information resulting from these analyses are published in Sneddon & Randall (1993b) for Wales and (1994a, b) for England and Scotland. The classificatory tables derived from the Twinspan analyses and their descriptions are published in Sneddon & Randall (1993a). There are 25 sections into which the communities are grouped based on 6 major categories (pioneer, secondary pioneer, mature grasslands, grasslands, heath and scrub). In all 124 separate communities (SH1 - SH124) are recognised. The closest match to an NVC category is also shown for each community; though in some cases there is no good match. [A summary of the main types is provided in Table 08-01 below. Details of each of these communities are given in the main report (Sneddon & Randall 1993a)].

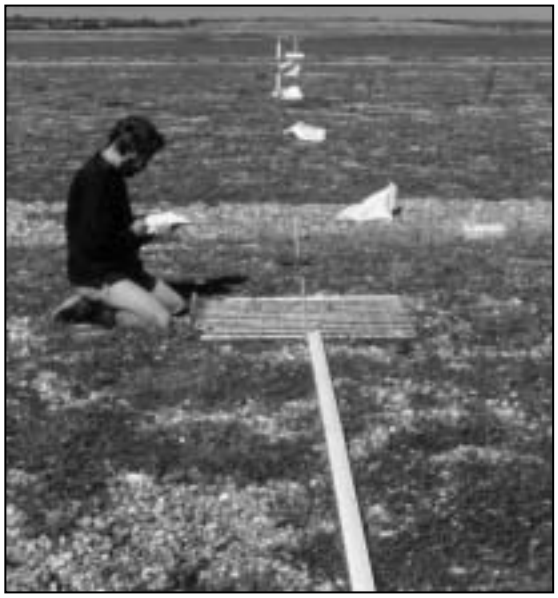
Table 08-01 Major divisions of the shingle vegetation classification. Divisions are listed in order broadly from the most landward to the most seaward vegetation types.

1. Scrub communities	la. *Prunus spinosa* communities lb. *Rubus fruticosus* communities lc. *Ulex europaeus* communities	
2. Heath communities	2a. Wet heaths 2b. Dry heaths	2b.i. *Pteridium aquilinum* 2b.ii. *Calluna vulgaris* communities 2b.iii Moss-rich communities
3. Grassland communities	3a. Saltmarsh-influenced grasslands 3b. *Agrostis stolonifera* grasslands 3c. *Arrhenatherum elatius* grasslands 3d. *Festuca rubra* grasslands 3e. Mixed grasslands 3f. Sandy grasslands	
4. Mature grassland communities	4a. Mature grasslands	4a.i. Mature grasslands - *Festuca rubra* 4a.ii. Mature grasslands - *Dicranum scoparium* 4a.iii. Mature grasslands - *Arrhenatherum elatius*
	4b. Less mature grasslands	4b.i. Less mature grasslands pure shingle 4b.ii. Less mature grassland saltmarsh influence
5. Secondary pioneer communities		
6. Pioneer communities	6a. *Honckenya peploides* dominated communities 6b. *Senecio viscosus* dominated communities 6c. *Beta vulgaris maritima* dominated communities 6d. *Raphanus maritimus* dominated communities 6e. Herb-dominated pioneer communities 6f. *Silene maritima* dominated pioneer communities	

Monitoring coastal vegetated shingle

The above techniques provide base-line surveys against which changes that may be occurring can be identified. Where change is identified or where management requires more specific monitoring to assess the effectiveness of treatment more detailed work may be required. Vegetated shingle is a designated habitat for HAP and several shingle species are recognised as Key Species in the National Species Action Plans. All sites should be recorded and monitored with these designations in mind. Some examples of specific monitoring techniques applied to coastal vegetated shingle are summarised below.

Permanent Transects and Quadrats



Where it is felt important to monitor change in a particular location within a site, permanent, relocatable quadrats may be set up. These can be placed randomly or systematically along a transect (Figure 08-01). Fuller & Randall (1988) assessing conflicts in coastal conservation used this latter method at Orfordness. In this study 10 relocatable vegetation transects were positioned to include all 5 shingle plant communities previously recorded and 45 10 x 10 m quadrats were recorded at the middle point of each vegetation zone. Presence/absence and species' percentage cover were recorded along with pebble size, litter type and disturbance. Ridges and lows were treated separately. Subsequently a continuous ribbon of 2 x2 m quadrats was run across the widest part of Orfordness spit to record the plant communities that were previously heavily disturbed within the old MOD fenceline.

Figure 08-01 Transect survey Orfordness using quadrats. The location of the transect (3.5) is shown in Figure 08-02 below.

This work helped to elucidate the essential conditions for plant growth on the 'inhospitable' shingle substratum. All these can be the basis of long-term monitoring programmes. Permanent sample plots must have their location marked on the ground or tied in to some permanent marker that can be relocated with accuracy. 'Permanent' may mean a season, a year or even a decade or more. Permanent quadrats may be constructed as 'exclosure quadrats' if grazing interference is to be avoided.

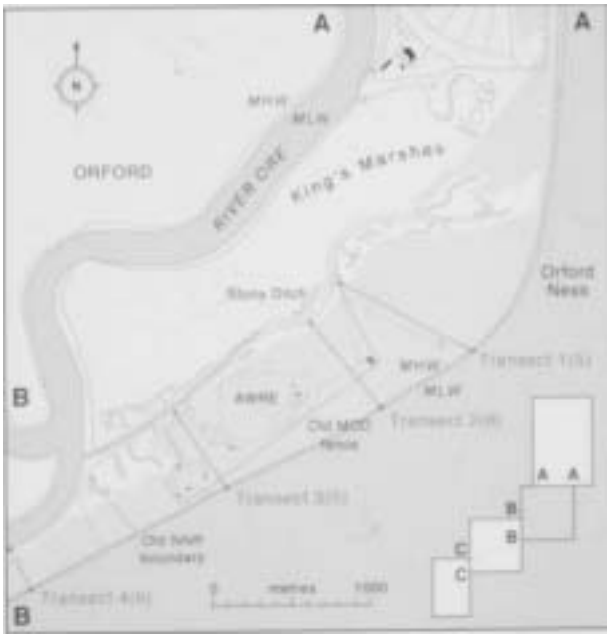


Figure 08-02 The location of transect 3.5 on Orfordness

Fixed point photography

A useful monitoring technique is to set up a series of locations from which to take repeatable photographs on a seasonal basis on approximately the same dates each year. These can be tied in with

permanent quadrats or transects. The key to all these approaches lies in the consistency (location, lens focus, direction and height) of approach, the periodicity of photographs and the timescale over which these are taken. The real value of this approach lies in the timescale over which the photographs are taken - the longer the better. When combined with historical information and maps they can be invaluable.

Animal data

A variety of techniques are available to survey animals on shingle structures and beaches. Surveys of invertebrates and birds can be of value in helping to determine management.

Invertebrate surveys

To date, most effort to understand the ecology of shingle systems has been confined to botanical evaluation. Detailed recording is largely confined to Dungeness and Rye Harbour (Morris & Parsons 1991, 1992). However, ad hoc surveys and collecting activity by a variety of recorders provide wider indications of the faunas of some shingle systems (Shardlow [2001] provides an overview). There is considerable scope to expand our understanding of the invertebrate assemblages on particular sites.

The majority of invertebrate groups are represented on shingle structures, but some offer better opportunities for comparative studies and to put individual sites into context. These include the spiders, beetles and bees, wasps and ants, which are all readily taken in static traps. The lepidoptera (moths and butterflies) are also well represented, but are mostly taken at light traps or through active searching for larval stages.

Shingle is, however, extremely difficult to sample. The most effective quantitative sampling on shingle is trapping using pitfall and water traps, but a very large number of replicates are needed to yield statistically significant results. This has important logistic implications and means that detailed surveys will be comparatively expensive. Invertebrate surveys are highly specialised and require strong taxonomic and field craft skills. General netting and searching does not readily yield many specimens, but can be useful to augment the results from static traps. A specialist with knowledge of what to look for will greatly increase the level of understanding of taxa with more specialist habitat requirements or those of a more retiring nature.

There are strong possibilities that southern shingle systems in particular will yield a range of important records. This is because they lie at the extreme of the continental range of many species and are highly thermophilic, favouring those taxa that can cope effectively with extreme temperature fluctuations. The nature of the shingle matrix, including levels of finer sediment within the matrix, influence plant communities and vegetation structure, which in turn has a bearing on the invertebrate assemblage.

Whilst invertebrate surveys often lead to the removal of substantial numbers of individuals, this is an unfortunate necessity. Many species can only be determined with confidence following dissection. If substantial numbers of a rare species are taken, consideration should be given to moving traps. However, the levels of trapping that are likely to be afforded are unlikely to significantly affect populations.

Bird surveys

Surveying birds can be undertaken using less specialist recorders. However, it is important that pitfalls in methods and study design are avoided. Bibby, Burgess & Hill (1992) have produced a detailed description of bird census techniques and have good examples of census forms for ‘seabird colony registers’ and transect techniques for 'apparently occupied nest sites'. Special methodologies for gulls and terns are described. For most colonial species that nest on shingle, a single count of the site in June provides a good estimation of numbers. However, repeat counts of dense colonies on different days are a valuable way of checking results. For resident or passage migrant species, monthly or annual bird counts should be carried out using standard methodologies. The important questions to ask in bird survey are:

- what is the purpose of the study;

- what field methods are affordable;
- what are the likely sources of bias;
- will the information give enough detail for analysis to work?

Gibbons, Hill & Sutherland (1996) detail the most useful answers to these questions and should be consulted before survey is undertaken in order to avoid the collection of unusable data. What is important about all species monitoring is that it will enable management to be carried out in relation to groups/species that are rare or seen to be suffering change rather than those that are merely spectacular!

What is important about all species monitoring is that it will enable management to be carried out in relation to groups/species that are rare or seen to be suffering change rather than those that are merely spectacular!

The UK Common Standards Monitoring programme

The above discussion is not exhaustive and other more specific prescriptions for monitoring coastal vegetated shingle are not available. Approaches which provide guidance on monitoring geomorphological characteristics are also not well documented. However, the Habitat Restoration Monitoring Handbook (Mitchley et al. 2000) provides a general, systematic approach to recording change, which is largely concerned with terrestrial areas. The UK government conservation organisations (English Nature, Scottish Natural Heritage, Countryside Council for Wales and the Environment Heritage Service in Northern Ireland) have been working on producing a Common Standards Monitoring (CSM) guidance for designated sites, under the direction of JNCC (more details on CSM can be found on the JNCC website). A summary of the guidance so far produced is given next.

Attributes and targets

A series of broad habitat attributes have been defined that should normally be part of the conservation objectives or the management plan for all sites where vegetated shingle is an interest feature.

There should normally be at least one target specified for each of the attributes. The targets set out here are for guidance only. They should be interpreted in terms of local knowledge of the site, its history and its surroundings. When a target is not applicable to a particular site it should be ignored, but a record of why the decision was taken should be made.

For vegetated shingle the mandatory attributes are:

- habitat extent;
- geomorphological naturalness;
- zonation of vegetation;
- vegetation composition for each vegetation zone;
- negative indicators (negative indicator species and signs of disturbance).

The presence of rare species (vascular plants) is considered to be a discretionary attribute. It will not be appropriate to use these ‘quality indicators’ on every vegetated shingle site, but where they are part of the reason for notification of the feature they should form an integral part (mandatory) of the condition assessment.

Guidance is given in the following sections as to what needs to be considered for the above attributes and, where appropriate, some examples are provided of the sorts of targets that should be set.

Recommended visiting period and frequency of visits.

The suggested visiting period is April to August, although annuals, such as *Arenaria serpyllifolia* will flower earlier and perennials e.g. *Crambe maritima* later and where driftline annuals are more abundant this will need to be taken into account.

Monitoring methods

Preparation

Prior to going out in the field collate existing information on your site. Aerial photographs (and possibly satellite images, see below) are particularly useful. Some NVC information should be available for most sites. Sneddon and Randall carried out a comprehensive survey in 1993 (see above) but other information may exist for individual sites. The use of the oblique aerial photographs in the DEFRA Futurecoast CD-ROM also provide an important record to allow familiarisation with difficult to access sites and to help place the feature in the context of the coastal environment.

Guidance: Read the guidance prior to the field visit, to familiarise yourself with what is required. You will need to tailor the assessment forms to your site.

Habitat extent should be assessed using any previous information available, preferably aerial photographs. If none is available this first reporting round must form the baseline. The source of the baseline must be clearly identified - aerial photography should include source, date (at least month and year) and scale. Field trials have shown that failure to provide some of this information may mean change cannot be measured with respect to the first round.

The assessment of zonation by transect should be done after the structured walks to assess sward composition and negative indicators of each vegetated shingle habitat as at that stage you will be more familiar with the vegetation at the site (see below).

In the field

It is recommended that the vegetation composition, negative indicators etc. of the two principal vegetated shingle 'zones' (annual vegetation of driftlines and perennial vegetation of stony banks) be assessed using a **structured walk** (e.g. a W shaped walk) with at least 10 stops within each assessment unit (block, management unit etc.) to avoid excessively variable results. The number of stops should be enough to allow the assessor to have an overview of the site and judge the condition of the feature. To avoid subjectivity in selecting stops and to ensure that as wide an area as possible is covered general routes with stops should be pre-selected based on a map or aerial photograph before the field visit. This also allows the number of stops per unit area to be determined more consistently. The **exact** stopping locations will be recorded in the field using GPS if possible. If contractors are using the guidance then consultation with local staff on route selection and stopping points is **mandatory**.

At each stop, the appropriate attributes (e.g. percentage cover and/or presence of relevant species) should be assessed within approximate 4 m^2 sampling units. There is no need to measure cover values precisely – simple visual estimates will suffice. It should not take very long (no more than 5 minutes) to record all the relevant attributes at each 'stop'.

The recommended methods of selecting the number and location of the stops are not intended to have statistical value, and the final condition of the interest feature is not simply the average of the condition of each stop. On the contrary, each stop should contribute to improve the assessor's overview of the state of the site. The following is a quantitative definition of frequency, intended to assist with the assessment of several of the saltmarsh attributes. This is a version of the well known DAFOR scale, which has been adapted to the particular characteristics of vegetated shingle as follows:

- **Dominant:** the species appears at most (>60%) stops and it covers more than 50% of each sampling unit;
- **Abundant:** species occurs regularly throughout a stand, at most (>60%) stops and its cover is less than 50% of each sampling unit;
- **Frequent:** species recorded from 31-60% of stops;
- **Occasional:** species recorded from 11-30% of stops.
- **Rare:** species recorded from up to 10% of stops.

Having completed the rapid assessments for the quality of the main vegetated shingle 'zones' it should then be relatively straightforward to do the one or more **transects** required to assess the width of the saltmarsh zones. If there is no prior data on the width of these habitats for your site, this assessment can form a crude baseline.

The routes followed for the structured walk and the start and finish points for the transects, should be marked on a map for future comparative use. Ideally these should be traced over aerial photos of the site using GIS, to enable comparisons on future visits. Permanent markers could be used to aid orientation in this respect. Comparisons could be made using set bearings from a given permanent marker – the owner/occupier could be asked not to destroy or remove particular makers (pill boxes etc) that could be used (although loss due to erosion will have to be accepted). Some landmarks may already be used as permanent markers for surveys by local authorities, the Environment Agency or universities.

Photographs are essential to the condition assessment and should be taken as an accompanying record wherever possible. These should be archived with the assessment file. In some countries photography is a mandatory part of the condition assessment.

There are several technologies involving remote sensing, which are being trialled to aid the condition assessment process, such as CASI and LIDAR (see below). These may provide a very useful tool for assessing zonation as well as extent.

The reporting unit

The assessment should be applied to the reporting unit, which is at present an SSSI site unit. However, where there is erosion in one unit and, following the key for extent shown below, it was assessed as unfavourable, it could be judged favourable **provided** it is certain that there is at least equal accretion in other units on the same site.

Habitat extent

Extent of the vegetated shingle is a fundamental attribute to be assessed in determining condition of the vegetated shingle feature. The target is no decrease in extent from the established baseline with the caveat 'subject to natural change'. There is a need to ensure that natural processes govern the system, and that the geomorphological 'health' of the feature is considered. However, the extent of the vegetation may be subject to periodic and seasonal variation, as natural processes re-distribute shingle sediments and change sediment composition. Extent of the driftline may be particularly difficult to assess as the community can vary enormously from year to year. The sparse nature of the vegetation and its seasonal appearance makes it difficult to detect on aerial photographs, but remote sensing images may be available. Driftlines on essentially sandy beaches are assessed within the sequence of sand dune communities. **The aim is to identify anthropogenic from natural processes which may be causing the feature to move away from favourable condition.**

Shingle deposits are dynamic and will adjust and respond to climatic changes (such as a rise in sea level) or local changes in wind and wave energy in an attempt to reach 'geomorphological equilibrium', which may include landward transgression. If this response is constrained by

anthropogenic constructions such as fixed sea defences or infrastructures, natural habitat migration is prevented ('coastal squeeze'). The ensuing loss of extent would fail the extent attribute, as the feature is prevented reaching a natural geomorphological equilibrium. However, if landward migration is prevented by a feature such as higher ground, this would be considered as favourable for the extent attribute (the shingle deposits are free to respond and reach a natural geomorphological equilibrium). The shingle structure might breach in response to ambient and storm effects, which would constitute 'roll over' - literally over landward habitat types including saltmarsh or sand dune.

The amount of offshore sediment available will also determine response of the system to such changes. There are other activities which may affect sediment supply, such as dredging see Chapter 03.

Zonation

The aim of this assessment is to detect negative trends arising from anthropogenic influences which are causing a long term decline in the habitat. One indicator involves assessing changes in the natural zonation of the habitats. Changes should be interpreted with great care given the dynamic nature of the vegetated shingle feature and in particular the driftline.

Vegetated shingle can usually be broken down into annual vegetation of driftlines and perennial vegetation of stony banks; however the NVC only describes part of the pioneer phase of perennial shingle vegetation, namely SD1 *Rumex crispus – Glaucium flavum* shingle community. Where a site has other communities, such as grassland, heathland or scrub these will need to be included in the overall assessment of zonation at the site.

Zonation of the vegetated shingle at some key points, using transects, will be assessed. The location of the transects is fixed by GPS, which means they can be easily relocated for future assessments. If a baseline is not available from an existing NVC survey, the condition assessment will form the new baseline. GPS information should be collected and marked on a map, and this should enable the width of the zones to be estimated.

Vegetation composition

After stability, the development of shingle vegetation is dependent on water availability, which is primarily controlled by the proportion and size of the fine fraction material of the matrix under 2mm in diameter. The matrix is usually composed of sand, silt or organic matter. The fine fraction is critical at germination and seedling stages, since without it enough moisture may not be present for growth to be initiated or to continue. The presence of finer particles of sand or detritus is essential for the development of the extensive absorptive roots most shingle plants need to thrive.

The greater expanse and stability on the largest and most stable structures results in a more complex development of vegetation over time. The sequence of vegetation may include long-lived perennials such as broom *Cytisus scoparius*, gorse *Ulex europeus*, and blackthorn *Prunus spinosa*. Heath vegetation with heather *Calluna vulgaris* and/or crowberry *Empetrum nigrum* can also occur, particularly in the north. The sequence of plant communities is influenced by natural cycles of degeneration and regeneration of the shrub vegetation that occurs on some of the oldest ridges. Sneddon and Randall (1991) gave a comprehensive description of stable shingle vegetation (see Table 02-01 above).

Vegetation composition - indicators of negative trends

There is a wide potential variation in the vegetation on more stable shingle banks, and invasion by many undesirable species is possible. Possible species occurring could be *Cirsium vulgare*, *Senecio jacobea* or *Pteridium aquilinum*; even gorse *Ulex europeus* can become a problem if the proportion becomes too high (e.g. over 10 % at Ballyquintin Point in County Down), although this would need to be done on a site-by site basis. *Lupinus arboreus* and *Tamarix gallica* are other introduced species on vegetated shingle.

Transitions to terrestrial habitats

A variety of terrestrial habitat types, such as heathland, grasslands etc. may be part of the vegetated shingle feature. However, transitions to other habitats such as saltmarsh, sand dune or M28 *Iris pseudacorus* mire may be present. Targets should be set according to conservation objectives or the management plan.

Other negative indicators:

In addition to observations on negative indicator species other negative indicators should be noted such as:

- signs of disturbance;
- vehicle damage or trampling at vulnerable locations (tracks, access points).

Indicators of local distinctiveness

Quality indicators are features of a shingle that make it 'special' but which are not covered by the attributes already described. They should be apparent from the SSSI citations or past surveys. This is a discretionary attribute in that it may not be applicable to every site; but where local distinctiveness has contributed to the selection of a site for shingle it should be mandatory. The target(s) should be tailored to each site. Quality indicators may include the following:

- notable species which are not notified features in their own right;
- structural attributes;
- associations between vegetated shingle and other habitats, e.g. mosaics of vegetation types, transitions to heath or freshwater reedbed.

Recording field forms

The countryside agencies will supply separate field recording forms that fit within the objectives detailed in this guidance.

It is advisable to record as much information as resources and time allow in a consistent manner during different visits and to keep all the records in a file. This will provide a track of the history of the condition of the site in relation to management. The route of the 'W' walk and transects should be marked on a map and photographs taken, especially of the more dynamic parts of the habitat.

The tables provided are for guidance only. Lists of species should be produced on a site-specific basis, and the assessment should be carried out based on the particular conservation objectives tables or management plans.

Use of satellites and other remote sensed images

Land based surveys are time consuming, especially where national data are being collected. A survey of saltmarsh vegetation in Great Britain (Burd 1989) took eight years to complete. Monitoring saltmarsh change in southeast England between 1972 and 1988 (Burd 1992) was expensive and time consuming. Neither study is likely to be repeated using traditional survey techniques. Other methods of large scale, more frequent survey would be more appropriate. In this context satellites may provide a possible alternative.

In the UK this approach has been initiated by English Nature and the Environment Agency National Centre for Environmental Data and Surveillance (NCEDS). The study has used combined LIDAR and CASI data collected by airborne instruments. These results need to be further investigated and extended.

CASI - **C**ompact **A**irborne **S**pectrographic **I**mager is a sensor which detects visible and near infrared electromagnetic energy reflected from the surface of the earth. The images provide information on ground-cover types to a high resolution. The imagery can also be located with a high degree of accuracy using a Global Positioning System (GPS).
LiDaR - **Li**ght **D**etection **a**nd **R**anging - is an airborne mapping technique using lasers to measure the distance from the aircraft to the ground. The results are presented in the form of a terrain map. It also provides information on coastal erosion and geomorphology.

The recent launch of ENVISAT should help provide an overview of the coast. There are several projects testing the value of this system and in time more use may be made, for example, in surveying and monitoring change at the coast. When combined with lower level airborne instruments a powerful tool could be developed. There is already calibration of remote sensed images for saltmarshes and it should easily be possible to use this approach for other coastal habitats. The issue needs to be addressed by a concerted effort, which brings together those who have knowledge of the coastal environment and those with technical expertise in remote sensing and GIS.

'Coastwatch'

The European Space Agency (European Space Research Institute) has recently let a service contract to a consortium led by the European Aeronautics Defence and Space Company (Systems and Defence Electronics) to provide an Environmental Monitoring Service for European coasts called 'Coastwatch'. The programme will focus on management and monitoring of the coastal zones of Europe. The area will include the land and the sea and provide information on specific coastal indicators, changes in land-cover, monitoring of erosion states, water condition monitoring and sea state dynamics. The policy sectors will include the Water Framework Directive, the Habitats and Species Directives and the Common Agricultural Policy. The approach taken will include the whole coastal zone, whilst at the same time responding to specific and multi-sectoral needs and providing information for user groups at all administrative levels. It is an ambitious project, starting in mid 2003.

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Chapter 09 Conclusions and knowledge gaps

The first eight chapters of the report provide a summary of the existing knowledge of the coastal vegetated shingle habitat, giving guidance on habitat management and restoration. This is largely based on existing information, supplemented by the reviews of casework at the six sites (Annexes 01-06) visited by the contractors together with the additional sites (Annexes 07-09). This section of the report provides some general conclusions. It also attempts to assess the extent to which current practices have or will help meet the targets identified in the UK Vegetated Shingle Habitat Action Plan (**HAP**). Based on this assessment knowledge gaps are identified, including those involving monitoring the impact of flood management schemes and the effectiveness of habitat enhancement/creation works. It begins by looking at the wider information needs before considering the more specific requirements for individual sites and species.

Habitat loss versus 'coastal' change

Chapter 05 provides a broad indication of the losses, which have been suffered by vegetated shingle beaches and structures as a result of human action. It also highlights the important management issues that need to be addressed when attempting to conserve the habitat and its associated species. Habitat Action Plan Target 1 (opposite) is now much more likely to be achieved as the protective legislation, notably strengthening the SSSI designation and the 'Habitats' Directive is more strictly enforced. For some of the most important sites, e.g. Orfordness and Dungeness ownership and / or National Nature Reserve status should help ensure that further major losses do not occur.

HAP Target 1 - "*Prevent further net loss of existing vegetated shingle structures totalling about 5800 ha. (However local gains and losses due to storm events occur sporadically and should be accepted provided that the national and regional resources are maintained overall.)*

The target also recognises the fact that natural events will cause changes to shingle features as erosion and deposition takes place. It would seem that a thorough review of this situation is needed to establish the 'context' in which the shingle features exist and potential long term declines in extent due entirely to natural changes. This may be particularly important in areas where sea level is known to be rising relative to the land.

The importance of understanding context

It is clear from the discussions so far, that the information obtained from the site studies when augmented by Futurecoast and the Executive Summaries for those CHaMPs overlapping with the study sites (Dungeness, Annex 05a, Orfordness, 06a and in relation to the Cley-Salthouse realignment Annex 07) that for many, if not most shingle beach and structure systems, sediment supply is limited. At some of the sites there may be no new sediment at all and the structures are relict features deriving from material deposited during the last Ice-Age. This certainly appears to be the case for two of the sites studied for this contract at Porlock (Annex 01) and in Start Bay Slapton (Annex 02). At the other sites on the south coast of England and at Orfordness although new material from cliff and shore platform erosion is available, the quantities appear to be insufficient to maintain the integrity of the systems. In several instances human intervention appears to have been instrumental in exacerbating erosional trends (e.g. Start Bay, Annex 02).

Coastal Cells

In recent years the concept of 'Coastal Cells' has developed in the UK. These represent a series of interlinked systems where sediment is moved around the coast by waves and currents (sediment transport cells). The cells and sub-cells that have been identified for England and Wales and Scotland comprise an arrangement of:

- sediment sources (e.g. eroding cliffs, river, sea bed);
- areas where sediment is moved by coastal processes;
- sediment stores or sinks (e.g. beaches, estuaries and offshore banks).

Together these have been used to define practical subdivisions of the coast (Figure 09-01) important for developing policy for Shoreline Management Plans. The Government has promoted the formation of voluntary coastal defence groups around these coastal cells made up of maritime district authorities and other bodies with coastal defence responsibilities. As part of this approach, DEFRA (2001) has issued guidance for maritime district authorities to help in the preparation of Shoreline Management Plans (SMPs) see http://www.defra.gov.uk/environ/fcd/policy/CoastalGroups.htm.

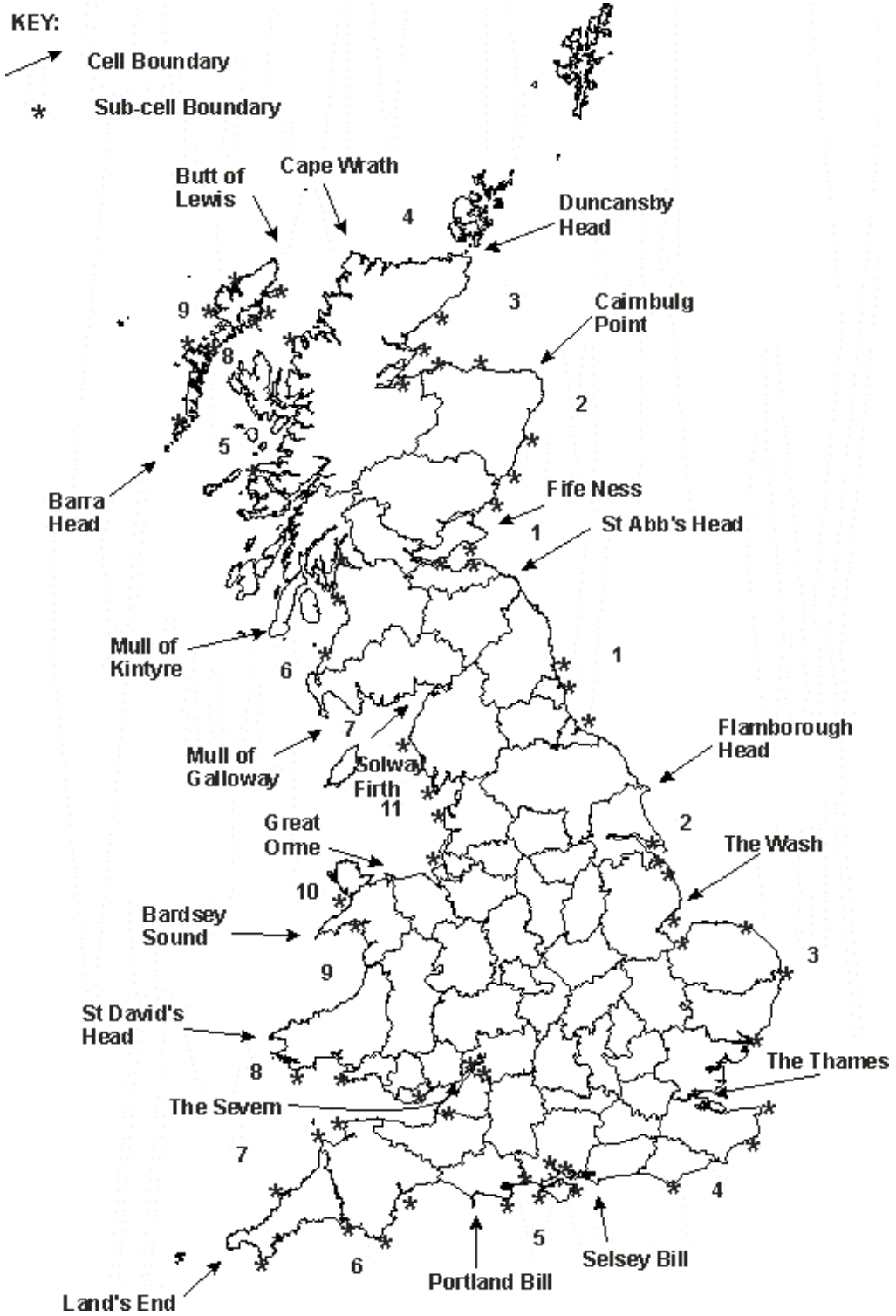


Figure 09-01 Coastal Cells and sub-cells in Great Britain.

Futurecoast

The Department of Environment, Food and Rural Affairs (DEFRA) and the National Assembly for Wales have recently collaborated on the promotion of a coastal process and geomorphological study of the coastline of England and Wales, called Futurecoast (see http://www.defra.gov.uk/environ/fcd/research/futurecoast.htm). The results of this project (Halcrow 2003) have recently been made available to selected organisations such as English Nature. Essentially it provides a linear view of the coastline of England and Wales at or around High Water. Included within it are assessments of the progression of the coastline based on maps of the location of the shoreline at different dates (derived from Ordnance Survey and Hydrographic Office data). These provide an indication of the way in which the coast is likely to react under different scenarios specifically in relation to the five generic policy options adopted by DEFRA as part of the development of Shoreline Management Plans.

Coastal Habitat Management Plans - CHaMPs

Of the 14 areas identified for the development of ChaMPs, 7 were chosen as pilot projects under a LIFE Nature funded project with English Nature, the Environment Agency and DEFRA.

The Executive Summary for each of these projects has recently been made available on the WEB (see EN 'Living with the Seas' project http://www.english-nature.org.uk/livingwiththesea/default.htm). These provide excellent overviews of the key factors determining the direction of the evolution of the coast and the likely consequences of adopting different shoreline management policies for each CHaMP. Those covering the study site reviewed for this contract are included as Annex 05a, 06a and 07a of this report.

Near-surface patterns of sea-water movement

The Ministry of Agriculture, Fisheries and Food (MAFF) (1981) produced a map of nearshore water movements (Figure 09-02) which goes some way towards explaining the five regional sectors that Sneddon (1992) discovered within the major community types of British shingle vegetation (Figure 09-03). More studies are needed on the importance of coastal currents in seed dispersal of shingle species, but enough is already known to highlight the importance of shingle vegetation management within the context of these regional sectors.



Figure 09-02 General near-surface pattern of water movement around the coast of the UK (after MAFF 1981).



Figure 09-03 Regional sectors for vegetated shingle vegetation in Great Britain derived from the analysis of plant community data. Note there is no eastern section because of the absence of anything other than isolated, small shingle deposits along this stretch of coast.

The importance of context

It is clear from the above that here has been an increasing use of the knowledge derived from taking a wider view of the management of the coastal zone and for the understanding of shingle structures and their associated communities.

Recommendation: These broad geographical zones should be used to provide sectors for the further assessment of change and the conservation needs of coastal vegetated shingle. The results of Futurecoast, the CHaMPs and the new generation of SMPs are valuable sources of information. In this context the recent comprehensive Regional Environmental Assessment of the cumulative impacts of planning applications to dredge offshore from the Sussex coast by the companies involved (Chapter 03 page 29) could prove to be an important initiative.

Armed with this **contextual** information decisions on future coast defence policy and planning issues such as siting of coastal development, will be better informed.

The reasoning behind this recommendation, from a biodiversity perspective at least, lies in the several ways in which such information can be used, namely:

1. helping to tease out human versus ‘natural’ evolution of the coastal shingle systems and with it the assessment of compliance with the HAP Target for the conservation of vegetated shingle habitat;
2. identifying and assessing potential areas where it would be appropriate for the development of new or restored habitat by allowing coastal processes of erosion and/or accretion to take place;
3. identifying areas where it would be inappropriate for ‘nature to be allowed to take its course’;

4. providing a link between the formation of the shingle structures and the development and management needs of the vegetation;
5. setting British shingle systems, and foreseeable changes to them, in a European and a World context.

This approach could be taken forward and included in the review of opportunities for biodiversity enhancement within 'natural areas' being undertaken by English Nature in the context of the development of the second generation Shoreline Management Plans. It also provides an essential component of all policy reviews for coastal areas and could be promoted through the EURosion Project (see http://www.eurosion.org/index.html), which is attempting to provide policy guidance for the European Union on coastal management of dynamic shores.

Sediment budgets

Determining sediment budgets is an important part of understanding the implications of a particular series of events or the likely outcome of human actions of coastal vegetated shingle. These studies have included work in both the coastal and nearshore environments as well as further offshore.

Coastal erosion studies

Some studies have attempted to provide sediment budgets for coastal systems as far dispersed as the Channel coast (the BERM project, southeast England has been described in Chapter 03) and Spey Bay in northeast Scotland (Annex 07). The former attempts to identify the contribution of contemporary material, derived from eroding cliffs to the integrity of the shingle beaches, whilst the latter takes a more wide-ranging view including sediment derived from the hinterland and the relationship between coastal processes and human coastal defence activities. The work of the BERM project is being continued and expanded upon to include work on biodiversity through the INTERREG III Beaches at Risk (BAR) Project.

Offshore sediment studies

The movement of sediment offshore has been studied by HR Wallingford in the "Southern North Sea Sediment Transport Study, Phase 2". This work looked at the implications of offshore dredging for marine aggregates on the marine environment. However, although not part of the remit of the study it recognised the importance of considering in detail… "*the possible physical effects of offshore dredging with particular regard to how such operations might alter waves, tidal currents, sediment transport and morphological changes of the seabed and along the coast. The results from such an assessment can be used to determine the likely effect on beaches and coastal defences*." (Information available from HR Wallingford web site where the results of the study can be downloaded @ http://www.sns2.org/project-outputs.html). At a regional scale (Brampton & Evans 1998) provide insights into the way sediment moves offshore.

It seems clear that managing coastal shingle deposits whether for sea defence or nature conservation, must take into account the contemporary contribution of material to the coast. Since this will be derived from or affected by the action of tides, winds and waves it is not only important to know how much is coming from erosion of the land, but also from the sea. The studies mentioned above provide some indication of the way in which the systems operate. However the link between the land-based and marine sources has not been made.

Recommendation: The implications of offshore dredging to the stability of the coast are implicit in the recommendations made for the revised Government View procedure and the production of an independent 'Coastal Impact Study' recognises this importance. Wide-ranging reviews of the sediment budget should be undertaken of the inter-relationships between offshore sediment movement and coastal change (erosion and accretion) in all areas where marine aggregate extraction is, or could take place. These should include offshore deposits and their relationship with onshore contemporary contributions to beach volume, for example derived from eroding cliffs.

The unpredictable

It is important to realise that understanding the inter-relationship between sediment budgets, the patterns of sediment movement and coastal change, is only part of the information need. Coastal process studies help to elucidate the nature and patterns of movement and the role they play in determining the geomorphic and ecologic patterns on coastal shingle. However, these processes can be severely disrupted by unpredictable and unforeseen events such as storms and storm surges. Having established the broad context for individual sectors of the coast it is suggested a vulnerability assessment could be made at progressively more detailed scales, including the coastal sub-cells.

Recommendation: Consideration should be given to providing a comprehensive and wide-ranging review based on existing sources of information (including Futurecoast and the detailed studies associated with SMPs and CHaMPs) of the vulnerability of shingle coasts to storms and other unpredictable events. This would contribute to the ongoing debate taking place at the European level through the EU funded EURosion project. [EURosion is attempting to identify at a European scale areas sensitive to erosion and/or flooding by virtue of their elevation, physical makeup, geomorphological characteristics and the rates of sea level change or incidence of storms. This information will in turn be used to identify appropriate policy responses, given the needs of the coast i.e. whether it is economically or environmentally significant.]

Control of visitors and other local activities

The site casework and general knowledge of the effect of localised recreational activities on shingle is well documented. In this study the situation at Rye and Pagham Harbours are particularly informative. Although there are many methods for controlling access and preventing other forms of damage it would appear that communication of the importance of shingle and the reasons for conserving the habitat, remains a key requirement for this habitat when considering meeting HAP Target 2, opposite.

HAP Target 2 - "*Prevent, where possible, further exploitation of, or damage to, existing vegetated shingle sites through human activities.*

Communications and publicity

It is extremely important to increase public awareness of the value and fragility of vegetated shingle through on-site interpretation. Although this has been carried out at some sites, to a limited extent, there is still much to be learned concerning the psychology of the use of public information notices what to say, how to say it, where to put car parks, interpretive boards etc. The reaction of British people is such that too terse information such as 'keep off' or 'no entry', which might be effective in Germany or Scandinavia, has little effect. Detailed information tends to put off the average visitor who only requires limited information, which is to the point. The examples of Nelson in New Zealand and Suffolk, England provide two contrasting examples which illustrate the point (Figures 09-04, 09-05 below).



Figure 09-04 Nelson Harbour Board sign, New Zealand, contains little information, is difficult to read and 'unsightly'. The sign also has a negative slant "NOT removing any boulders…"; "NOT taking vehicles onto the beach…."; "NOT taking dogs…"

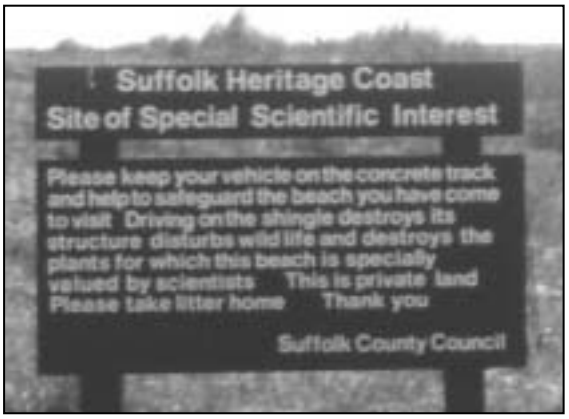


Figure 09-05 Heritage Coast sign at Shingle Street, Suffolk.

Awareness raising video

The Nature Coast Project and East Sussex Coastal Biodiversity Project have produced a short awareness raising/educational video on vegetated shingle, for use both on site in visitor or field study centres or in a wider area for example in schools, training events and public presentations. Both projects also put a considerable emphasis on raising awareness of vegetated shingle (and other coastal habitats) through events, guided walks, talks to local groups etc.

Recommendation: A more concerted national effort is required to raise the profile of shingle areas both among the general public and those with responsibility for the management of vegetated shingle sites. The initiatives taken by the West Sussex Vegetated Shingle project provide a possible model.

Comment: *In response to the production of the Sussex coastal vegetated shingle HAP, a partnership project was set up in 1999 to help achieve some of the action plan targets, in particular raising awareness of vegetated shingle in West Sussex. A project officer was employed for 3 years during which time a variety of methods were used to raise the public profile of vegetated shingle, including producing a leaflet, a mobile display, website, education pack, resources for coastal defence engineers, interpretation boards, a programme of presentations and enabling community participation. The project is acknowledged as having been successful in what it set out to achieve.*

Management

HAP Target 3 - *"Maintain the quality of existing plant and invertebrate communities which are currently in favourable condition.* ***Revised****"*

This study, particularly the site reviews, has highlighted the extent of management activity already being undertaken on many important sites in Great Britain. The ability to manage habitats and species, as well as people, within these areas is in many instances well tried and tested. For some sites (Pagham, Annex 03; Rye, Annex 04 and Orfordness, Annex 06) there are readily and easily accessible sources of information on the management techniques which have been adopted. However, this study has only been able to look at six sites (Annexes 01 - 06) due to time and cost constraints. Cley-Salthouse (Annex 07) has been included because it highlights many of the issues surrounding adopting a limited intervention approach to coastal defence. Culbin (Annex 09) and Spey Bay (Annex 08) were included to provide examples of a relatively untouched site and one where regeneration of disturbed / excavated shingle has taken place, respectively. The maps provided in Chapter 04 show there are many other vegetated shingle sites in Great Britain (there are also many more in Ireland) and a wealth of further information to be collected and collated on management strategies.

One of the primary aims of this contract was to provide management guidance for vegetated shingle commensurate with that available for the other primary coastal formations namely sand dunes, saltmarshes, sea cliffs, coastal wetlands and grazing marsh. However there remain many aspects of the management of coastal vegetated shingle that have only been covered in a relatively superficial way.

Recommendation: In order to fill in the gaps in knowledge both geographically and in more depth a further study should be carried out. This should look in more detail at specific issues such as grazing management, significance of vegetation control in relation to ground-nesting birds and control of recreational activities (see also below invertebrates).

Invertebrates

Invertebrates are one of the least well known groups of British shingle species, yet where studies have been carried out the habitat has been shown to hold considerable numbers of Red Data Book (RDB) and priority Biodiversity Action Plan species. A species of fly new to science was recently found living within the shingle at Rye suggesting there is much more to discover. Invertebrates are not only important in their own right as part of the biological diversity of shingle areas, but they are also important indicators of the health of the systems. Although there is detailed information for a few sites (notably Dungeness) it is clear that many other sites are under recorded. At the same time there is a paucity of knowledge on the importance of habitat type, the association with host plants and interactions between the two. Although some plants are known to be particularly important for some species, many more rely on a matrix of habitats including open and closed shingle vegetation. Invertebrates may also have a close relationship with and depend upon the physical structure of individual plants and/or plant communities. Thus the architecture of vegetation and the shingle matrix is very important. Not understanding this relationship runs the risk that management such as grazing, if advocated on largely stable vegetation, could lead to loss of invertebrate interest.

Recommendation: It is vitally important to extend the knowledge of the interactions between habitat, plant species and community structure to those sites that are currently understudied. These studies should be carried out across the range of shingle habitats, from bare shingle to closed grasslands, heathlands and scrub. The work should be designed to develop management plans that incorporate the protection of invertebrate taxa and their microhabitats. These studies should be linked to objectives associated more directly with vegetation management. There are strong possibilities that southern shingle systems, with their wealth of species lying at the edge of their geographical range, will yield a range of important records and could be tackled first.

Ecology of shingle drift lines (beaches).

At the time JNCC funded research on coastal vegetated shingle structures in Great Britain a decade ago, it was realised that no research had been carried out on the shingle foreshore habitats (strandlines) or on maritime islands. However, these are prime locations for the ‘Habitats’ Directive Annex1 Habitat: Annual Vegetation of Drift Lines. A study of shingle beaches was carried out for Dumfries and Galloway (Randall 2001) but this remains the only recent work of this kind.

Recommendation: It is vital that funding is found to carry out work on the ecology and conservation of this habitat, which is currently poorly understood.

Comment: *As has already been suggested the National Vegetation Classification only describes part of the pioneer phase of perennial shingle vegetation, namely SD1 Rumex crispus – Glaucium flavum shingle community, which at present is used to describe the 'Perennial vegetation of stony banks' Annex I habitat (1220) type.*

Patch dynamics and the establishment of vegetation

Work carried out at Kingston (Annex 08) has suggested that the abundance and diversity of lichens and particularly bryophytes indicate a mechanism of initial colonisation and primary succession on bare terrestrial shingle that has not been studied elsewhere. The resultant 'islands' of vegetation offer a model of succession on bare shingle with no maritime influence, a situation unusual in Britain but more common in Baltic Germany, Sweden and Norway.

Recommendation: Undertake successional studies of shingle island vegetation at Kingston, Scotland.

Restoration

There are several programmes which are attempting to restore vegetation to shingle areas on several of the sites reported in this study. The principle approaches involve the use of plant material. However, these are few in number, at a small scale and of limited duration. Regrading shingle in conjunction with the introduction of plant material has also been tried but again on a very limited scale. There are clearly opportunities for extending experimental work in both these areas.

HAP Target 4 - "*Achieve the restoration, where possible, of degraded or damaged habitats of shingle structures, including landward transitions, where such damage has been extensive and natural recovery is not likely to be initiated, by 2010. **Revised***"

Seeding and planting

A lack of experimental data on the ability of shingle beach species to prevent erosion or repair damaged shingle has meant that no large scale planting of native species has occurred. Walmsley and Davy's work (Walmsley & Davy 2001) at Sizewell has been reviewed in Chapter 07 and current experiments at Pagham and Dungeness are reported in Annex 03 & 05, respectively.

Recommendation: Undertake more experimental work on plant re-establishment using different substrate matrices, seeding combinations and different precipitation regimes.

Transplantation

Where loss of mature vegetated shingle is inevitable because of decisions to build on stable habitats, consideration might be given to transplanting existing vegetation. There are no examples reported so the effectiveness of treatment is not known. Evidence from the sites where seeding has been tried suggest that this would not be easy.

Recommendation: Consider, **as a last resort**, experimental transplantation of mature shingle in areas where destruction is inevitable.

Regrading shingle in damaged areas

Little work has been carried out to date on geomorphic restoration of damaged shingle-ridge systems. The work at Orfordness (Annex 06) is extremely valuable if inconclusive. Recent work at Rye Harbour (Annex 04) has shown that buried systems can, with care, be excavated to display significant geomorphic sequences once more. More work should be carried out in this area to assess the value of more extensive restoration of the physical landscape.

Recommendation: Consideration should be given to a large-scale re-grading experiment using an industrial plant in an existing extraction site. This could/should be accompanied by experiments to see how distance from the water table and the introduction of fine fractions influence the rate of colonisation and type of communities that are formed.

Plant autecological studies

Although several of the important plant species of shingle have been written up for the Biological Flora of the British Isles *e.g. Suaeda vera* (Chapman 1947), *Mertensia maritima* and *Glaucium flavum* (Scott 1963) and *Crambe maritima* (Scott & Randall 1976) - http://www.britishecologicalsociety.org/publications/journals/ecology/biologicalflora.php, there is still need for much greater understanding of the ecology of many other species. Some detail is available from the Ecological Flora of the British Isles database at York University - http://www.york.ac.uk/res/ecoflora/cfm/ecofl/index.cfm but it would be very useful to have more knowledge on the autecology of species such as the shingle ecotypes of *Cytisus scoparius, Galium aparine, Geranium robertianum*, *Prunus spinosa* and *Solanum dulcamara* or species like *Crithmum maritimum, Eryngium maritimum* or *Tripleurospermum maritimum.*

Recommendation: Sponsor more autecological studies of rare plants and those potentially providing added stability to shingle beaches.

Letting nature take its course

Geomorphology

Over the years coastal defence works at Orfordness have been extremely damaging to the ecology of the area. A breach in the spit at Slaughden meant that 250,000 m^3 of shingle was transported from the Ness to fill the gap (Kinsey 1981). The King's Marshes were also bulldozed towards the shore to raise the sea defences. Both courses of action restricted the width of the beach, such that a minor breakthrough could quickly become much worse. Just below Aldeburgh, the river has cut into the landward side of the spit, while the sea erodes the outer shore because defence works further north starve the shoreline. Holding the line is only possible by beach feeding. It would be very helpful to carry out detailed modelling to ascertain the affect of ceasing beach feeding and leaving this site to nature. Would losses be that great? Is the present practice sustainable? Steers (1964) pointed out that the current situation at Orfordness is quite similar to that at Dungeness in the Middle Ages. Left to itself the Alde would eventually break through as the Rother once did. This might well be a cost-effective solution to a sea defence problem that would allow the coast to develop naturally without great loss or cost to man or the natural environment.

Recommendation: Undertake predictive modelling on the geomorphological implications of leaving certain sites to nature e.g. ceasing beach feeding at Slaughden (Orfordness, Annex 06).

Monitoring at Porlock

There are relatively few contemporary examples of situations in which natural processes have been allowed to re-create reprofiled shingle ridges and Porlock (Annex 01) is probably the prime example in the UK. It is increasingly being argued that in some cases allowing ‘nature to take its course’ may be a much more sustainable and cost-effective option to maintain coastal and flood defence than continuing to adopt the engineered solution.

Although detailed studies have taken place on the geomorphological change, monitoring the vegetation has been undertaken at a relatively small scale and by local staff. The opportunity exists (notably at Porlock, Annex 01 and possibly at Orfordness, Annex 06 and Cley-Salthouse, Annex 07) to undertake detailed monitoring both of the physical and ecological changes that ensue and the relationship with the socio-economic environment. Added to this is the possibility of seeking the views of the local population and assessing changing attitudes to taking this less interventionist approach. This case has already been used by the Field Studies Council as a case study for A-Level and GCSE Geographers (McTernan & Wilson 1999) and poses many of the questions relevant to this debate.

Recommendation: A detailed monitoring strategy should be set up for Porlock encompassing both the changes to the shingle ridge itself and the development of vegetation and animal usage of land now open to tidal inundation. This should be further developed to investigate attitudes to the socio-economic consequences of this less interventionist approach.

Letting people know things may not be as bad as they think.

Because there is a history of many centuries of winning land from the sea and creating strong sea defences, there is a natural aversion of people living in coastal areas to any reversal in this policy. There is a general feeling that any let-up in defences in the light of global warming, sea level rise and land subsidence in the south and east, will result in massive losses of coastal land (especially low land protected by shingle). With the evidence of current examples of managed realignment (dealt with in more detail in the “Good Practice Guide for habitat restoration, re-creation and creation on the coast”,

it is important to highlight that 'letting nature take its course' may be a reasonable option where current defences are unsustainable in the long term. The case of Porlock also suggests that 'things may not be as bad as they seem' when 'natural' realignment of the coast takes place.

Recommendation: Take a more proactive approach to promoting the environmental and socio-economic benefits of taking a less interventionist (more realistic) approach to shingle barrier breakdown. This should include an explanation of how the coast will change in response to sea level rise/climate change.

Shingle-sand and shingle-saltmarsh ecotones

The implications of shingle restoration for habitats lying inland of the shingle beach/structure when 'roll-over' is contemplated, have already been discussed in relation to Porlock (Annex 01) and Cley (Annex 07). That there will be changes in the nature of the habitats is certain, with marginal habitats most likely to be replaced and a consequential overall loss of biodiversity.

Brightmore (1979) and Adam (1990) highlight the fact that there is a vegetation community comprising characteristic species such as *Frankenia laevis, Limonium bellidifolium, L. binervosum* and *Suaeda vera*, which occurs in a specialised habitat where saltmarsh, sand dune and shingle are in juxtaposition. This assemblage represents a Mediterranean element in the British flora and is best developed on the north Norfolk coast. However, a related, less diverse community is present in similar locations on the Sussex coast and in Anglesey. This ecotonal habitat is narrow, linear and susceptible to minor environmental changes and has declined markedly in recent times.

Recommendation: Augment the phytosociological understanding of these rare and valuable series of transitional community types with more detailed studies of their ecology. In this way it should be possible to be more certain of the likely consequences of taking a non-interventionist approach as one community is replaced by another. It will also help provide the basis for predicting the necessary environmental conditions, which might favour the re-creation of transitional habitats when managed realignment schemes are considered.

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Chapter 10 Funding sources and useful contacts

There are a number of different funding sources for management and restoration of vegetated shingle. Most of these are not specific to the habitat but the schemes most commonly used are identified below. These are presented from EU level downwards.

European Union

Several European Union sources of funding have been used for shingle related projects. LIFE, which is the Financial Instrument for the Environment, introduced in 1992, is one of the main means through which the European Union environmental policy is progressed. It co-finances projects in two areas relevant to shingle management and restoration:

1. **LIFE Nature** - involves actions aimed at conservation of natural habitats and the wild fauna and flora of European Union interest, according to the Birds and Habitats directives. They support implementation of the nature conservation policy and the Natura 2000 Network of the European Union;
2. **LIFE Environment** - involves actions which aim to implement the Community policy and legislation on the environment in the European Union and candidate countries. This approach enables demonstration and development of new methods for the protection and the enhancement of the environment;

Two other schemes potentially provide support for restoration and other work:

3. **Agri-environment** funding mechanism (agri-environment regulation, Council Regulation No (EEC) 2078/92) provides for programmes to encourage farmers to carry out environmentally beneficial activities on their land. The programmes are managed by regional or national authorities under a decentralised system of management, subject to approval by the Commission for each programme. The costs are part-financed from the EU budget - 75% in objective 1 areas and 50% elsewhere;
4. Interreg III follows up Interreg II (see below page 102). It is an initiative which aims to stimulate interregional cooperation in spatial planning in the European Union during the period 2000-06 and is financed under the European Regional Development Fund (ERDF). [Two projects relating specifically to shingle were carried out under INTERREG II, and a further shingle project is being undertaken under INTERREG III (Beaches at Risk).]

Each of these mechanisms is described further below where reference is made to the value of each for funding shingle related projects.

LIFE Nature

LIFE Nature is an appropriate mechanism for any shingle site which is included in the Natura 2000 series (candidate Special Areas of Conservation and Special Protection Areas). There are two projects with funding under this mechanism affecting vegetated shingle habitats. These are:

1. **The conservation of Orfordness** (LIFE94 NAT/UK/000850)

Having purchased the site in 1994, the National Trust looked towards the LIFE fund to help implement the actions needed to help restore the site. This work involved activities including re-establishing damaged or destroyed biotopes, re-introducing grazing regimes aimed at encouraging the return of Annex I species ('Habitats' Directive) and controlling many damaging activities like hunting and the use of four wheel drive vehicles. The project has provided the basis for the long term conservation of the site.

Duration: 1st January 1995 to 1st March 1998.

2. **Wild Ness : the conservation of Orfordness, Phase 2** (LIFE97 NAT/UK/004245)

This second project has built upon the successes and results of the first project in an attempt to restore the site to a favourable conservation state. In specific terms, the project identified the factors

suppressing the breeding of Annex I ground nesting birds and the development of rare shingle flora. It has acted upon these findings to improve grazing on saltmarsh, coastal grazing marsh and along the river wall. New lagoons have been created for breeding and feeding waders. Restoring damaged shingle surfaces has also been attempted. The management plan has been updated and revised and provides a long term conservation/ restoration programme for the site.

Duration: 1st March 1997 to 1st April 2000

A summary of the results of these projects is given in the site report for Orfordness (Annex 06).

Living with the sea

This is also a LIFE Nature projects which aims:

To promote:

- understanding of long term (30-100 year) coastal change resulting from sea level rise;
- sustainable integrated coastal management policies;
- ownership of shared issues and common solutions.

To develop:

- mechanisms for delivering Habitats Directive compliant flood and coastal defence schemes e.g. CHaMPs;
- practical ways of demonstrating habitat creation at work e.g. North Norfolk Flood Defence schemes;
- working partnership between engineers, conservationists and landowners.

The project includes the Guide "Coastal Habitat Restoration, towards good practice".

For **further information** on the LIFE programmes and projects see the **LIFE** home page at (http://europa.eu.int/comm/environment/life/home.htm) which provides information on:

- the programme in general terms and the three components (a third element is for Life - third countries);
- what's happening within the programme, for example upcoming LIFE events, selection of new projects, other opportunities etc;
- practical steps when applying for funding - annual deadlines, application forms, who to contact etc;
- the results and lessons of the LIFE programme;
- past, present and future projects;
- all the operators involved in the day to day work on the programme;
- contacts at all levels, such as with European institutions, Member States, candidate countries, external teams, Mediterranean and Baltic Sea countries, etc.

Life Environment

There are no shingle related projects funded through this mechanism. However the LIFE Regulation defines five areas eligible for funding:

- land-use development and planning;
- water management;
- reduction of the environmental impact of economic activities;
- waste management;
- reduction of the environmental impact of products through an integrated product policy.

For more information see http://europa.eu.int/comm/environment/life/life/environment.htm.

Agri-environment Schemes

Examples of the type of land management activities carried out include:

- reversion of intensively used land, such as arable or grass for silage to biologically diverse, but unprofitable extensive grassland;
- reduction in use of nutrients (resulting in loss of yield);
- reduction or cessation of use of pesticides (e.g. organic farming);
- creation of nature zones taken out of production;
- continuation of traditional environmental land management in zones liable to neglect;
- maintenance of landscape features which are no longer agriculturally viable.

Information can be obtained via EC web page @
http://www.europa.eu.int/comm/agriculture/envir/index_en.htm

DEFRA operates two principle schemes: Countryside Stewardship and Environmentally Sensitive Areas (ESAs) under the European Union Agri-environment Scheme. Details of these are provided below:

Countryside Stewardship

Countryside Stewardship Schemes are governed by the Countryside Stewardship Regulations 1998, as amended, made under Section 98 of the Environment Act 1995. The scheme forms part of the England Rural Development Programme, part-funded by European Union funds, and is therefore subject to Council Regulation 1257/1999, Commission Regulation 1750/1999 and Commission Regulation 2075/2000. The scheme is administered by the Department of Environment Food and Rural Affairs (DEFRA) in consultation with the Office of the Deputy Prime Minister, Countryside Agency, English Heritage and English Nature. It offers payments to farmers and landowners to improve the natural beauty, habitats and wildlife of the countryside and to help people to enjoy them. The schemes are managed through a 10 year legally binding agreement, the maximum period “normally” permitted under Regulation 1257/1999. It operates outside Environmentally Sensitive Areas. Anyone who owns or manages suitable land may apply. The scheme is open to farmers and non-farming land owners and managers, including voluntary bodies, local authorities and community groups.

The Scheme is discretionary and the more work of different types entered the better. DEFRA gives priority to plans which amongst other things:

- include land in a county target area;
- are of landscape, wildlife and historical interest;
- are linked to land of special interest or contribute to a national target for a habitat or species listed in a Biodiversity Action Plan;
- offer a combination of different benefits;
- provide opportunities for people to enjoy the benefits;

- cover land in a village or urban fringe of high local amenity value.

Grants are also more likely if plans represent a positive change in management, are realistic and achievable, will produce the planned objectives, are well researched, are supported by environmental bodies and are good value for money. **NB**: Works associated with diversion of public rights of way are not eligible for Countryside Stewardship funding.

From January 2000, the Stewardship scheme has included an intertidal habitat creation option and payments for capital work where appropriate. The specific objectives for coastal areas relevant to coastal shingle are:

- re-create intertidal habitats on agricultural land;
- manage intertidal habitats including saltmarsh and **shingle ridges** where changes or active management are required.

Information on who can apply, how to submit an application and the levels of payment can be found on the DEFRA web site via the England Rural Development Home Page @ http://www.defra.gov.uk/erdp/erdphome.htm). [A review is currently underway to examine the future use of these payments, follow the link to the DEFRA Agri-environment Review.]

Environmentally Sensitive Areas (ESA)

Environmentally Sensitive Areas are specially designated areas where payments are made to enhance the conservation, landscape and historical value of the key environmental features of an area, and, where possible, improve public access to these areas. There are currently 22 ESAs in England. Each has a series of tiers identifying operations for which payments can be made to farmers. For example, Broads, Suffolk River Valleys, North Kent Marshes and Somerset Levels have tiers relating to raising water levels and maintaining wet grassland.

Comment: *None of these schemes currently have particular value for shingle habitats, though where roll-over or breaches to shingle structures are contemplated management of the hinterland could be enhanced if the land lies within an ESA.*

Wildlife Enhancement Scheme (WES)

English Nature operates the Wildlife Enhancement Scheme giving financial support to land managers in achieving positive management on SSSIs by means of short term agreements of up to five years. Standard payments may be made for achieving appropriate nature conservation management by means such as grazing, hay cutting, or water control over wet grassland. They may also be made for carrying out specific management works such as hedge laying, tree planting and scrub or bracken control or managing coastal wet grassland such as at Pevensey Levels. So far there are no examples on shingle areas, though this funding source could be relevant.

CCW's Berwyn Scheme

The Countryside Council for Wales administers a similar scheme - the Berwyn Standard Payment Scheme, which includes measures to ensure compatibility between farm management practices on hill grazings and the land's international, ornithological and habitat importance. Payments are made on a per hectare basis to farmers and land managers who agree to adopt agreed management programmes which regulate seasonal grazing and undertake rotational heather burning and pest control. Where appropriate, additional payments can be made to assist other management, such as bracken control and the planting of native tree species. There is no known use of this initiative for shingle habitats.

Interreg II

The Interreg Community Initiative, which was adopted in 1990, was intended to prepare border areas for a Community where internal national frontiers are of less significance. The aim of the Regen

Initiative launched in the same year was to help fill in some of the missing links in the trans-European networks for transport and energy distribution in the Objective 1 regions. It had three distinct strands, namely:

- Interreg II A (1994-1999): cross-border co-operation;
- Interreg II B (1994-1999): completion of energy networks;
- Interreg II C (1997-1999): co-operation in the area of regional planning, in particular management of water resources.

Two projects with interests relating to shingle were:

1. **BERM** - Beach Erosion in the Rives Manche

This set out to investigate the supply and depletion of shingle on Rives Manche beaches under present day management regimes, and to determine the sustainability of the beaches in a period of rising sea level. This project was an Interreg II project with matching funds coming from the relevant local authorities. [An Interreg II programme funded by the European Union (Priority E: Environmental Conservation and Enhancement. Measure 9: Information, Prevention and Awareness)]. The study was undertaken by the Coastal Research Group, University of Sussex and the Université de Rouen and Université de Caen. It will help determine the extent to which contemporary shingle beaches of Rives-Manche are depleting and assess the risks that this will pose for the region in the face of rising sea levels.

2. **Two Bays, One Environment Project** Interreg II East Sussex and France

The purpose of the project was to compare the wildlife and habitats of the TWO BAYS (Rye Bay East Sussex and Baie de Somme, northern France) with a view to improving management techniques for the rare species and habitats. The 50% match funding was provided by the East Sussex County Council.

The work included creating a joint database. Species records have been entered onto a biological database. This also involved the collection of new data and the exchange of surveyors and specialists. The project also provided an opportunity to undertake joint studies of wildlife management techniques with an exchange of experience between environmental workers. Habitat management advice was also given to farmers and landowners within the project areas.

Another important aspect to the project was to promote the understanding of the environmental importance of the area to decision makers, farmers, landowners and the general public. This took place through leaflets, displays, newsletters, guided walks, interpretation boards, workshops and a conference. Many of the publications were bilingual, and both areas displayed information about the project and its partner across the channel. The Communication and Interpretation Plan identifies the core and subsidiary messages presented to the public. The project includes Rye Harbour (Annex 04), which is one of the two sites included in the Interreg project - see also web site @ http://home.clara.net/yates/2bays.html).

Interreg III

Interreg III follows up Interreg II. It is an initiative which aims to stimulate interregional cooperation in the European Union between 2000-06 and is financed under the European Regional Development Fund (ERDF). This new phase of the Interreg initiative is designed to strengthen economic and social cohesion throughout the EU, by fostering the balanced development of the continent through cross-border, transnational and interregional cooperation. Special emphasis has been placed on integrating remote regions and those which share external borders with the candidate countries.

Again programme III is made up of 3 strands, with an emphasis on cooperation across regional and national boundaries:

Strand A: cross-border cooperation between adjacent regions aims to develop cross-border social and economic centres through common development strategies. (Only the southeast of England, Wales and eastern Ireland are eligible within the UK and Ireland.);

Strand B: transnational cooperation between national, regional and local authorities aims to promote better integration within the Union through the formation of large groups of European regions;

Strand C: interregional cooperation aims to improve the effectiveness of regional development policies and instruments through large-scale information exchange and sharing of experience (networks).

Supplementing the three strands, networks promoting the sharing of experiences and best practice will be funded. See http://europa.eu.int/comm/regional_policy/interreg3/index_en.htm for further information.

Interreg III is delivered through a series of programmes. These programmes set the regional priorities for funding. The opportunities for 'shingle' related projects depend on the provisions of the programming documents.

Interreg Programmes currently operating in the UK for the period 2000-2006 are:

- Interreg IIIA Franco- British Programming Area;
- Interreg IIIB North Sea Area Programme;
- Interreg IIIB Atlantic Area Programme;
- Interreg IIIB North West Europe Programme;
- Interreg IIIC West Zone Programme.

There may be particular opportunities for projects that relate to aspects of shingle protection and management from the North West Europe Programme and the North Sea Programme.

The North West Europe Programme may fund trans-national projects under the priorities:

- water resources and the prevention of flood damage;
- stronger ecological infrastructure.

The North Sea Programme may likewise fund trans-national projects under the priorities of:

- sustainable management of the Environment, natural Resources and Cultural Heritage, in the areas;
- creative rehabilitation, protection and development of cultural and natural landscapes and townscapes;
- integrated and concerted management and planning of coastal zones and the North Sea itself.

INTERREG III Project Beaches at Risk (BAR).

The two English Partners are the University of Sussex and East Sussex County Council. Aims to continue and expand upon the work carried out through BERM on beach sediment budgets, and the East Sussex Coastal Biodiversity Project carried out under Interreg II. Further information is available at http://www.geog.sussex.ac.uk/BAR/.

Countryside Agency grants

Funding is available from each of the countryside agencies:

English Nature Aggregate Levy

This fund includes opportunities for funding by English Nature through the Aggregates Levy Sustainability Fund, as is the case of shingle restoration at Rye Harbour. Grants are also available from the principle countryside agencies, namely:

Countryside Council for Wales (details from their web site http://www.ccw.gov.uk/);

Scottish Natural Heritage (details from their web site http://www.snh.org.uk/);

Northern Ireland Office (details from their web site http://www.nio.gov.uk/).

Flood and coastal management

Responsibility for funding sea defence schemes lies with the Department of Environment, Food and Rural Affairs. It gives financial support to the flood and coastal management operating authorities in England (these are the local authorities, the Environment Agency and Internal Drainage boards) for between 35% and 80% of the value of capital projects. Several of the schemes discussed here (Cley-Salthouse, Annex 07) and the majority of those on the south coast of England are funded in this way. In some instances DEFRA will fund such schemes which also enhance Natura 2000 sites damaged by human actions (see for example Freiston Shore in the Wash, Living with the Sea “Good Practice Guide for habitat restoration, re-creation and creation on the coast” and Cley-Salthouse Annex 07a).

Other sources

These include the National Heritage Lottery Fund, which supports the Nature Coast Project, which helped produce the video described above (page 92).

Chapter 11 Glossary of terms

Abiotic	without life; an absence of living organisms
Alien	not native, introduced to a region deliberately or accidentally by man and now ± naturalised
Allogenic	Succession where replacement of one community by another results mainly from environmental changes
Anastomosing	dividing up then joining again
Apposition Beach	beach that has grown in area as a result of successive deposition of ridges over time
Autecology	the study of individuals or species in response to environmental conditions
Autogenic	succession wherein replacement of one community with another results chiefly from changes induced by the plants themselves
Backwash	the movement of seawater downslope after a breaking wave has taken it to the highest point up the beach
Capillarity	the ability of a liquid to rise up narrow channels
Clast	rock material broken down to any sediment size
Climax	The plant community which, in a given area, was thought to perpetuate itself indefinitely
Closed-turf communities	species rich plant and animal communities that develop over a relatively long time, gradually building up a layer of soil and humus over the shingle, eventually completely covering it with vegetation, with no bare shingle visible.
Coastal defence	a combination of both Coast Protection (generally to prevent erosion – where the land is higher than sea level) and Sea Defence (to prevent flooding- where the land is lower than sea level).
Coastal squeeze	the shrinking of coastal habitats caught between rising sea level and fixed coastal defences.
Distal	the point furthest from origin
Dominant	the chief constituent of a particular plant community
Ecosystem	the biotic community and its non-living environment as an interacting system
Ecotone	a transition zone between two adjacent and differing communities
Edaphic	that which relates to the soil
Endemic	referring to a group of organisms native to a given region
Environment	all the conditions and influences surrounding and effecting an organism or community
Escape	a plant growing outside a garden derived from cultivated specimens
Flora	the total assemblage of plant species present in an area
Habitat	the specific kind of environment occupied by the individuals of a species
Halophyte	A plant that grows successfully in salty conditions
Holocene	The period of approximately the last 10000 years
Homeostasis	the maintenance of equilibria in an organism or system
Horizon	soil layer
Humus	dark brown or black organic matter in soils formed from the decomposition of dead plants and animals
Intermediate communities	plant and animal communities that are intermediate between pioneer and established, closed-turf communities.
Interpretation	means of displaying explanatory information for example in the form of a leaflet or notice board.
Introduced	not native, known to have been brought in by man within historic times
Leaching	the downward movement or removal of soluble chemicals from the soil
Matrix	the finer material present between shingle clasts
Nutrient enrichment	increase in nutrient content of shingle.
Oceanicity	the condition of being influenced by the physical attributes of the sea.
Phenology	study of periodicity in plants and animals
Pioneer communities	plant and animal communities that establish within a relatively short time and are usually the first to colonise a new area.
Quaternary	geological period of approximately the last two million years including the Pleistocene and the Holocene (the last 10000 years)

Ranker	a well-drained soil with an organic upper layer and little-altered, non-calcareous, unconsolidated material below
Relict	residual feature that has survived over a long period of time and is no longer geomorphologically active.
Ruderal	a plant characteristic of waste places
Sere	a group of plant communities that successively occupy the same site
Stochastic	statistically by chance
Succession	the changes in vegetation and animal life by which one population or community is replaced by another
Topography	characteristics of the ground surface in terms of physical features
Vegetated shingle	shingle supporting a community of flora and fauna.
Weed	any plant considered undesirable by man in a given location

Abbreviations

AONB	Area of Outstanding Natural Beauty
BAP	Biodiversity Action Plan
CCW	Countryside Council for Wales
CHaMP	Coastal Habitat Management Plan
COAST	Co-ordinated Action on Seaside Towns
cSAC	Candidate Special Area of Conservation
DEFRA	Department of the Environment, Food and Rural Affairs
EA	Environment Agency
EIA	Environmental Impact Assessment
EIA	Environmental Impact Assessment
EMP	Estuary Management Plan
EN	English Nature
ERDF	European Regional Development Fund
EU	European Union
EUCC	European Union for Coastal Conservation
GIS	Geographical Information Systems
GCR	Geological Conservation Review
HAP	Habitat Action Plan
ICZM	Integrated Coastal Zone Management
JNCC	Joint Nature Conservation Committee
LNR	Local Nature Reserve
NFU	National Farmers Union
Ramsar	Site designated as an International Wetland Site
RDB1	Red Data Book (endangered category)
RSPB	Royal Society for the Protection of Birds
SAC	Special Area of Conservation
SCOPAC	Special Conference on Problems Associated with the Coastline
SMP	Shoreline Management Plan
SNH	Scottish Natural Heritage

SPA	Special Protection Area (for birds)
SSSI	Site of Special Scientific Interest

Shingle and shingle related bibliography

[NB Individual references are provided at the end of each Chapter and in the Annexes; the following is a consolidated reference list and bibliography]

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APPENDIX 01 A SUMMARY OF LEGISLATION

[Produced by David Tyldesley MIEEM, FRTPI, FRSA for English Nature]

This Annex provides an overview of the various regulatory controls that may affect proposals to manage or restore shingle habitats. These fall into two principal categories:

- Controls that would relate to all such proposals, irrespective of site designations – general regulatory controls; and
- Controls that are specific to and apply in relation to particular designations – specific regulatory controls.

This chapter also looks at the key policy frameworks that may influence the decisions taken under these procedures and provides maps showing the location and extent in Great Britain of sites designated for their biological or geomorphological shingle interest features. These designations are principally Sites of Special Scientific Interest (SSSI), National and Local Nature Reserves, candidate Special Areas of Conservation (cSAC) and Ramsar Sites.

GENERAL REGULATORY CONTROLS

Development requiring planning permission

Development is defined by section 55 of the Town and Country Planning Act 1990 and section 26 of the Town and Country Planning Act (Scotland) 1997 as "*the carrying out of building, engineering, mining or other operations in, on, over or under land, or the making of any material change in the use of any buildings or other land*". Generally, planning permission is required to undertake any works that fall within the definition of development that are not exempt by the 1990 / 1997 Act. In the case of physical works to vegetated shingle habitats, involving the movement of shingle or construction of structures such as groynes, it is likely that planning permission will be required. Permission may also be required if a material change of land use would be involved, but such cases are likely to be infrequent. Planning permission is not required for ongoing management of habitats that do not involve engineering or mining operations.

Extraction of shingle for use as a mineral would be a mining operation and will always require planning permission. All but minor movements of shingle, even where there is no intention to use it as a mineral, are likely to be an engineering operation in the meaning of the Act and will also require planning permission.

Planning permission for engineering and mineral operations on shingle would normally be obtained following the submission and consideration of a planning application by the relevant planning authority. If the local planning authority refuses permission it may be granted, on appeal, in England and Wales by a Planning Inspector (acting under delegation from the first Secretary of State / Welsh Assembly Government) or in Scotland by a Reporter (acting under delegation from the Scottish Ministers).

Certain works, for example by the Environment Agency or Scottish Environment Protection Agency (SEPA) or a local authority, may be granted planning permission generally by the Secretary of State / Welsh Assembly Government, through the provisions of Article 3 and Schedule 2 of the Town and Country Planning (General Permitted Development) Order 1995 or by the Scottish Ministers through the provisions of Article 3 and Schedule 2 of the Town and Country Planning (General Permitted Development) (Scotland) Order 1992. These general grants of planning permission are referred to as "permitted development". An express planning application does not have to be made for permitted development because permission has already been granted. However, these permitted developments are subject to conditions, two of which are explained further below in respect of the Habitats Regulations and the EIA regulations.

Most new or replacement local authority coast protection schemes will require planning permission on an application. However, because the local authority is also the planning authority, such applications

are dealt with under special procedures in the Town and Country Planning (General) Regulations 1992 or Town and Country Planning (Development by Planning Authorities) (Scotland) Regulations 1981.

Sections 54A and 70 of the Town and Country Planning Act 1990 / S.25 and 37(2) Town and Country Planning (Scotland) Act 1997 require planning authorities (and the Secretary of State / Welsh Assembly Government and Inspectors and, in Scotland, Scottish Ministers / Reporters) to determine planning applications in accordance with the development plan unless material considerations indicate otherwise[2]. It means that the development plan is a particularly important consideration in deciding planning applications. The development plan consists of the Structure Plan and the Local Plan (including any Mineral and Waste Local Plans) or the Unitary Development Plan[3]. Other material considerations may include national planning guidance that comprises, in England, the series of Planning Policy Guidance Notes (PPGs) and the fifteen Mineral Planning Guidance Notes (MPGs) issued since 1992. PPG7 *The Countryside: Environmental Quality and Economic and Social Development* 1997; PPG9 *Nature Conservation* 1994; PPG20 *Coastal Planning* 1992 and PPG25 *Development and Flood Risk* 2001 are particularly relevant here. The series of PPGs and MPGs is currently under review and will be replaced by new government planning policy statements (PPSs).

Similar guidance is found in relation to Scotland in Scottish Planning Policy (SPP) 1, *The Planning System* 2002; National Planning Policy Guidelines (NPPG)7 *Planning and Flooding*, 1995, NPPG13 *Coastal Planning* 1997 and NPPG14 *Natural Heritage* 1999; and, in relation to Wales, in *Planning Policy Wales* 2002 and Technical Advice Notes (TAN) 5 *Nature Conservation and Planning* 1996, TAN14 *Coastal Planning* 1998 and TAN15 *Development and Flood Risk* 1998. In Scotland the series of NPPGs is being replaced by Scottish Planning Policies, SPPs.

The jurisdiction of planning control is normally down to MLWM (MLWMOST in Scotland).

Works below MHWM

Works or deposit of materials below MHWM (eg movement of shingle on the intertidal areas or construction of groynes) will require the consent of the Crown Estate Commissioners (or other landowner). A licence under the Food and Environment Protection Act (FEPA) 1985 may also be required from D*efra* and in some circumstances, where works below MLWS may affect navigation, consent from the Department for Transport (Ports Division), Welsh Assembly Government or Scottish Executive will be required under the Coast Protection Act 1949.

Works on or affecting flood or coastal defence structures

Any works to structures that form part of coastal protection schemes or flood defences, whether natural or artificial, will require the agreement of the Coast Protection Authority (usually the local authority) or the Environment Agency respectively. Policy and strategic planning for flood defence and coastal protection is embodied in D*efra*, 2001, *Shoreline Management Plans – A Guide for Coastal Defence Authorities* and the national series of Shoreline Management and Estuary Management Plans. Other D*efra* publications are accessible via www.defra.gov.uk/environ/fcd/pubs

Land drainage works

In this context land drainage has a wide definition. Works that may affect a Land Drainage Improvement Scheme authorised under the Land Drainage Act 1991 or Land Drainage (Scotland) Act 1958 or Flood Prevention and Land Drainage (Scotland) Act 1997 will need to be agreed with the appropriate land drainage body, likely to be the Environment Agency, Internal Drainage Board (IDB) or Scottish Environment Protection Agency (SEPA).

[2] This requirement is more fully explained in paras 39 – 56 of Planning Policy Guidance Note (PPG) 1 *General Policy and Principles*, 1997 (England), and paras 46 – 49 Scottish Planning Policy (SPP) 1, *The Planning System*, 2002.

[3] See PPG 12 *Development Plans* 1999; paras 3.1.1 – 3.1.6 *Planning Policy Wales*, 2002 and *Unitary Development Plans Wales* National Assembly for Wales 2001; and paras 25 – 40 SPP1, *The Planning System*, 2002).

Rivers and other watercourses

If the water level in a main river is likely to be affected by any works to the shingle, or if it is proposed to discharge into a watercourse, consent is required from the Environment Agency or SEPA. Consent from the local IDB will also be required for proposals likely to affect the water level of main drains or other watercourses on land in another ownership.

Environmental impact assessment

It is unlikely that most habitat restoration or management schemes for vegetated coastal shingle will be subject to the Environmental Impact Assessment (EIA) process. However, where undertaken in connection with flood defence or coast protection schemes, or other types of project listed in Schedules 1 or 2 of the Town and Country Planning (Environmental Impact Assessment) (England and Wales) Regulations 1999 or the Environmental Impact Assessment (Scotland) Regulations 1999 the whole project may be subject to EIA procedures. Works that may otherwise be permitted development (see above) cannot proceed without an application for express planning permission being made if they would constitute EIA development[4].

SPECIFIC REGULATORY CONTROLS

Works on or affecting a SSSI

Under the provisions of S.28 of the Wildlife and Countryside Act 1981, as amended, English Nature, CCW and SNH have a duty to notify Sites of Special Scientific Interest (SSSI) where they are of the opinion that the land is of special interest by reason of its flora, fauna, geological or physiographical (including its geomorphological) interest. The reasons for notification are known as the site's interest features. It should be noted that the provisions of the section in relation to England and Wales were modified and extended by section 75 and Schedule 9 of the Countryside and Rights of Way (CRoW) Act 2000 and differ substantially from the provisions in Scotland.

In England and Wales any operations likely to damage the interest features of a SSSI are subject to one of three types of control, under section 28E, 28H or 28I, depending on who would be carrying out the works, where, and whether they need the consent of a public body. The provisions are set out below in outline only and reference must be made to the relevant sections of the Act that prescribe the procedures to be followed and how offences may arise for non-compliance.

Section 28E

The operations likely to damage a SSSI are listed in the citation accompanying the notification of the site; examples relevant to shingle habitat conservation are given in the box below.

[4] Detailed guidance is provided on EIA procedures in DETR Circular 2/99 *Environmental Impact Assessment*;Welsh Assembly Government Circular 11/99 *Environmental Impact Assessment;* Scottish Executive Circular 15/1999 *The Environmental Impact Assessment (Scotland) Regulations 1999* and Scottish Executive Planning Advice Note (PAN)58 *Environmental Impact Assessment* 1999.

Some operations likely to damage shingle interest features of SSSI

- Grazing
- Reclamation
- Gravel extraction
- Use of vehicles or craft.
- Draining or water abstraction.
- Sea defences or coast protection works
- Reprofiling.
- Roads, tracks, walls, fences, hardstands, banks, ditches or other earthworks.
- Erection of permanent or temporary structures.

Before any operations that are likely to damage the interest features of the SSSI may be carried out, within a SSSI, by an owner or occupier, or by someone on their behalf, or with their permission, the owner or occupier must first give notice to EN / CCW and may not proceed with the works unless:

a) they have received the written consent of EN / CCW; or
b) the operations are carried out in accordance with a management agreement under previous legislation, or a management scheme (under S.28J) or a management notice (under S.28K); or
c) the works were an emergency operation and particulars of them (including details of the emergency) are submitted to EN / CCW as soon as practicable after the commencement of the operation; or
d) the operations have been authorised by a planning permission granted on an application (this excludes permitted development, see above) or in accordance with a consent granted by another public body (called a Section 28G Authority see below) that has followed the consultation requirements of S.28I (also described below).

Conditions can be imposed on any consent granted under S.28E. Further guidance on the requirements and implications of S.28E can be obtained from the EN website at www.english-nature.org.uk

Section 28G

Section 28G defines all public bodies and persons holding any public office as a "Section 28G Authority" and imposes a general duty on them to take reasonable steps, consistent with the proper exercise of their functions, to further the conservation and enhancement of the interest features of all SSSI.

Section 28H

Under S.28H, a Section 28G Authority must give notice to EN / CCW before carrying out any operations likely to damage any of the interest features of a SSSI, whether or not the operations would take place in the SSSI. If EN / CCW do not assent to the operations the S.28G Authority may only carry them out in accordance with procedures set out in S.28H and, in any event, shall carry out the operations in such a way as to give rise to as little damage to the interest features as is reasonably practicable in all the circumstances, taking account of any advice from EN / CCW. The authority must also restore the site to its former condition so far as is reasonably practicable.

Section 28I

S.28I provides for cases where a Section 28G Authority is responsible for giving any kind of consent or authorisation to operations likely to damage any of the interest features of a SSSI, whether or not the operations would take place in the SSSI. Before giving any permission in these cases, the S.28G Authority must give 28 days notice of the proposed operations to EN / CCW and must take account of any advice received. If the Authority proposes to grant permission to the operations against the advice of EN / CCW (or not including conditions recommended by EN / CCW) they must give EN / CCW notice of the terms of the permission, how they have taken account of the advice and ensure that the operations cannot commence until after the expiry of 21 days notice to EN / CCW.

Further guidance on the requirements and implications of S.28G - I can be found in DETR Circular 04/2001 *The Countryside and Rights of Way Act 2000.*

Sections 28E and G – I, as described above were introduced by the Countryside and Rights of Way Act 2000 and do not apply in Scotland. SNH does not have the power to refuse to permit operations likely to damage the interest features of SSSI and public bodies do not have the duties in relation to carrying out or permitting operations. However, new legislation is expected to be proposed in the Scottish Parliament. Emerging Scottish legislation can be monitored on www.scotland.gov.uk/legislation

Works on or affecting National and Local Nature Reserves

National and Local Nature Reserve (NNR and LNR) designations are a recognition of a site's particular national or local importance. The designation is a material consideration in decisions but these designations do not have associated legislation that imposes specific duties on decision makers, or others wishing to carry out operations or projects that may affect them. However, all NNRs and many LNRs will be SSSI so the above provisions will apply and, in any event, works in NNRs and LNRs should be consistent with the terms of any nature reserve agreement or other legal / management agreements in force. NNRs are particularly important for their potential contribution to research and education and are owned or otherwise managed by EN / CCW / SNH or managed by other owners / occupiers under specific NNR agreements with nature conservation as the priority. LNRs can only be designated by local authorities, in consultation with EN / CCW / SNH, under the provisions of S.21 National Parks and Access to the Countryside Act 1949.

Works on or affecting a European Site

In addition to the controls relating to SSSI, there are further controls imposed by Regulations 48 – 53 of the Conservation (Natural Habitats &c) Regulations 1994. Regulation 3 imposes a general duty on all competent authorities (widely defined in Reg.6 to include any public body or person holding a public office), in the exercise of any of their functions, to have regard to the requirements of the Habitats Directive so far as they may be affected by the exercise of those functions.

Specifically, Regulation 48 requires that, before deciding to undertake or to give any kind of consent, licence, permission or other authorisation, for a plan or project (widely defined to include any kind of new works), a competent authority must consider whether the proposal would be likely to have a significant effect on a European Site in Great Britain, either alone or in combination with any other plans or projects. If it is likely to have such an effect, it must be subject to an appropriate assessment. This is an assessment of the implications of the project in view of the conservation objectives for the site. Plans and projects directly connected with and necessary to the management of the site for nature conservation are exempt from the requirements for an assessment.

In England European Sites are all classified SPAs and cSAC, and in due course, Sites of Community Importance (SCI) and fully designated SACs. In Scotland and Wales they are at present only classified SPAs, but see below in respect of proposed European Sites and Ramsar Sites.

The competent authority must consult EN / CCW / SNH, and have regard to their representations, and may consult the general public. It must consider the way in which the project is proposed to be carried out and any restrictions such as conditions that may be imposed. The competent authority cannot grant permission or undertake the project unless:

a) It can ascertain that the project would not adversely affect the integrity of the site; or
b) If the project would adversely affect the integrity of the site or there is uncertainty over the possible effects,
    i. there are no alternative solutions; and
    ii. there are imperative reasons of overriding public interest (which are more restricted in scope if a habitat or species listed as a priority in the Habitats Directive would be affected).

If a competent authority is minded to grant a permission or undertake a project in the circumstances of (b) above, it must first give notice to the Secretary of State / Welsh Assembly Government / Scottish Ministers and comply with any directions they may issue. If permission is to be granted the government must ensure that any necessary compensatory measures are taken to ensure the overall coherence of Natura 2000 is protected[5].

Regulation 60 of the Habitats Regulations imposes a condition on all forms of permitted development (see above) that would be likely to have a significant effect on a European Site in Great Britain, either alone or in combination with other plans or projects. Such permitted development shall not be carried out or continued without the prior written approval of the planning authority. This does not in itself withdraw permitted development rights but subjects permitted development likely to affect a European Site to a prior approval procedure (under Regulation 62) in which the planning authority will undertake an appropriate assessment and consult EN / CCW / SNH. Permitted development directly connected with and necessary to the management of the European Site for nature conservation is exempt from the requirements for prior approval. If the planning authority can ascertain that the permitted development would not adversely affect the integrity of the European Site, it should approve the development which may then proceed as permitted development in accordance with the General Permitted Development Order. If the planning authority cannot ascertain that the permitted development would not adversely affect the integrity of the European Site, it is prohibited from granting prior approval and a full planning application would need to be made which would be subject to the requirements of Regulations 48 – 53 described above.

Works on or affecting a proposed European Site or Ramsar Site

In England candidate SACs are already European Sites as a matter of law. In Scotland and Wales cSACs are not European Sites as a matter of law but it is government policy to treat them and (as in England) proposed SPAs and classified Ramsar Sites as if they are fully designated European Sites for the purposes of considering development proposals that may affect them[6].

Works affecting a Scheduled (Ancient) Monument

If any proposed works would be undertaken on or adjacent to (i.e. affecting the setting of) a Scheduled (Ancient) Monument it would be necessary to obtain, in England a Scheduled Monument Consent from English Heritage; in Wales a similar consent from Cadw; and in Scotland a Scheduled Monument Consent from the Scottish Ministers (Historic Scotland) under the provisions of The Ancient Monuments and Archaeological Areas Act 1979[7].

[5] Further guidance on the application of these regulations can be found in Annex C of DoE PPG9, 1994; Ministerial Statement House of Commons, published by DETR, 12 May 1998, *Guidance on Application of Articles 6(3) and (4) of the Habitats Directive*; paras 39 – 45 of Scottish Office Development Department NPPG14 *Natural Heritage* 1999; Scottish Executive (SERAD) Circular of June 2000, *Nature Conservation: Implementation in Scotland of EC Directives on the Conservation of Natural Habitats and Wild Flora and Fauna and the Conservation of Wild Birds (the Habitats and Birds Directives)* and paras 6 – 20 Welsh Office, TAN5 *Nature Conservation and Planning* 1996; European Commission DG Environment, *Managing Natura 2000 Sites The provisions of Article 6 of the Habitats Directive 92/43/EEC*, April 2000 and European Commission DG Environment, *Assessment of Plans and Projects Significantly Affecting Natura 2000 Sites – Methodological guidance on the provisions of Article 6(3) and 6(4) of the Habitats Directive 92/43/EEC*, November 2001

[6] Further guidance on this policy may be found in paras 13 and C7 DoE PPG9, 1994; DETR, *Ramsar Sites in England A Policy Statement*, 2000; the footnote page 1 of Welsh Office TAN 5 *Nature Conservation and Planning* 1996; and para 39 Scottish Executive Development Department NPPG14 *Natural Heritage* 1999.

[7] Guidance on taking archaeological interests into consideration in developments is set out in DoE PPG16 *Archaeology and Planning*, 1990; Welsh Office Circular 60/96 *Planning and the Historic Environment: Archaeology*, 1996; and Scottish Office Environment Department NPPG5 *Archaeology and Planning*, 1994.

Works affecting a Listed Building or Conservation Area

It is possible that shingle areas may contain (or may lie in the curtilage of) buildings or other structures that are Listed Buildings of Special Architectural or Historic Interest on registers of Listed Buildings compiled by English Heritage, Cadw and the Scottish Ministers (Historic Scotland). Any works that may affect such buildings or structures will require Listed Building Consent in addition to any planning permission or other consents required, under the provisions of the Planning (Listed Buildings and Conservation Areas) Act 1990 or the Planning (Listed Buildings and Conservation Areas) (Scotland) Act 1997. In considering any such application, Ss. 16 and 66 of the 1990 Act require any planning authority, Inspector, the Secretary of State or the Welsh Assembly Government to have special regard to certain matters, including the desirability of preserving the setting of the building and any features of special architectural or historic interest it possesses as well as the building itself. A similar requirement is imposed on planning authorities, Reporters and the Scottish Ministers in Ss.14 and 59 of the 1997 Act[8].

Works impeding a public right of way

If works would obstruct a public right of way or impede passage along such a route it would be necessary to obtain a temporary (or permanent) diversion order via the local Highway Authority (England and Wales) or Roads Authority (Scotland).

Other Considerations

Other, general, legal considerations that could apply to such works are not covered in the above discussion but may include such matters as health and safety, employment law, human rights, compliance with any legal agreements (including tenancy, financial and management agreements) and / or restrictive covenants on land and obtaining the consent of private landowners.

[8] Guidance on Listed Building Consents and development control in Conservation Areas is provided in DoE PPG15 *Planning and the Historic Environment*, 1994; Welsh Office Circular Circular 61/96 *Planning and the Historic Environment: Historic Buildings and Conservation Areas*, 1996; Scottish Office Development Department NPPG18 *Planning and the Historic Environment*, 1999 and Scottish Office *Memorandum of Guidance on Listed Buildings and Conservation Areas* 1998.

Appendix 02 Descriptions of the shingle element within the candidate Special Areas of Conservation, derived from the Joint Nature Conservation Committee web site

Chesil Beach

1210 Annual vegetation of drift lines

Chesil Beach is a large (28km-long), relatively undisturbed shingle bar, and is one of two representatives of **Annual vegetation of drift lines** on the south coast of England. The inner shore of the beach supports extensive drift-line vegetation dominated by sea beet *Beta vulgaris* ssp. *maritima* and orache *Atriplex* spp. This community exists in a dynamic equilibrium with the perennial shrubby sea-blite *Suaeda vera* community typical of **1420 Mediterranean and thermo-Atlantic halophilous scrubs** (***Sarcocornetea fruticosi***), for which this site has also been selected.

1220 Perennial vegetation of stony banks

The 28km-long shingle bar of Chesil Beach, with the contiguous Portland Harbour shore, is an extensive representative of **Perennial vegetation of stony banks** on the south coast of England, and most of it is relatively undisturbed by human activities. Much of the shingle bar is subject to wash-over and percolation in storm conditions and is therefore sparsely vegetated. It supports the most extensive occurrences of the rare sea-kale *Crambe maritima* and sea pea *Lathyrus japonicus* in the UK, together with other grassland and lichen-rich shingle plant communities typical of more stable conditions, especially towards the eastern end of the site.

Dungeness

1210 Annual vegetation of drift lines

The Dungeness foreland has a very extensive and well-developed shoreline, although with sparse vegetation and in places some human disturbance. It is one of two representatives of **Annual vegetation of drift lines** on the south coast of England. The strandline community on this site comprises Babington's orache *Atriplex glabriuscula*, which occurs mostly on the accreting eastern shoreline, although it is also present on the eroding southern shoreline.

1220 Perennial vegetation of stony banks

Dungeness is the UK's largest shingle structure and represents the habitat type on the south-east coast of England. The total area of exposed shingle covers some 1,600ha, though the extent of the buried shingle ridges is much greater. Despite considerable disturbance and destruction of the surface shingle, the site retains very large areas of intact parallel ridges with characteristic zonation of vegetation. It still has the most diverse and most extensive examples of stable vegetated shingle in Europe, including the best representation of scrub on shingle, notably prostrate forms of broom *Cytisus scoparius* and blackthorn *Prunus spinosa*. A feature of the site, thought to be unique in the UK, is the small depressions formed within the shingle structure, which support fen and open-water communities.

Blakeney Point

1220 Perennial vegetation of stony banks

Perennial vegetation of stony banks occurs at Blakeney Point, a shingle spit on the east coast of England with a series of recurves partly covered by sand dunes. This extensive site has a typical sequence of shingle vegetation, which includes open communities of pioneer species on the exposed ridge and more continuous grassland communities on the more sheltered shingle recurves. It also includes some of the best examples of transitions between shingle and saltmarsh, with characteristic but rare species more typical of the Mediterranean. These include one of the best examples of the transition from sand and shingle to vegetation dominated by shrubby sea-blite *Suaeda vera* (**1420 Mediterranean and thermo-Atlantic halophilous scrubs**). Blakeney Point is part of a multiple-interest site. The shingle structure forms a highly significant component of the geomorphological structure of the North Norfolk Coast and helps to maintain a series of interrelated habitats.

Minsmere to Walberswick

1210 Annual vegetation of drift lines

This site is one of two representatives of **Annual vegetation of drift lines** on the east coast of England. It occurs on a well-developed beach strandline of mixed sand and shingle and is the best and most extensive example of this restricted geographical type. Species include those typical of sandy shores, such as sea sandwort *Honckenya peploides* and shingle plants such as sea beet *Beta vulgaris* ssp. *maritima*.

Orfordness

1210 Annual vegetation of drift lines

Orfordness is an extensive shingle spit some 15km in length and is one of two sites representing **Annual vegetation of drift lines** on the east coast of England. In contrast to Minsmere to Walberswick Heaths and Marshes, drift-line vegetation occurs on the sheltered, western side of the spit, at the transition from shingle to saltmarsh, as well as on the exposed eastern coast. The drift-line community is widespread on the site and comprises sea beet *Beta vulgaris* ssp. *maritima* and orache *Atriplex* spp. in a strip 2-5m wide.

1220 Perennial vegetation of stony banks

Orfordness is an extensive shingle structure on the east coast of England and consists of a foreland, a 15km-long spit and a series of recurves running from north to south on the Suffolk coast. This spit has been selected as it supports some of the largest and most natural sequences in the UK of shingle vegetation affected by salt spray. The southern end of the spit has a particularly fine series of undisturbed ridges, with zonation of communities determined by the ridge pattern. Pioneer communities with sea pea *Lathyrus japonicus* and false oat-grass *Arrhenatherum elatius* grassland occur. Locally these are nutrient-enriched by the presence of a gull colony; elsewhere they support rich lichen communities. The northern part of Orfordness has suffered considerable damage from defence-related activities but a restoration programme for the shingle vegetation is underway.

Culbin Shingle Bar

1220 Perennial vegetation of stony banks

Historically, Culbin Bar in north-east Scotland formed part of the same shingle aggregation as Lower River Spey – Spey Bay to the east. Although sea-level rise has separated the sites, they are still linked, being maintained by the same coastal processes. Culbin Bar and the Lower River Spey – Spey Bay are, individually, the two largest shingle sites in Scotland and together form a shingle complex unique in Scotland. They represent **Perennial vegetation of stony banks** in the northern part of its range in UK. Culbin Bar is 7 km long. It has a series of shingle ridges running parallel to the coast that support the best and richest examples of northern heath on shingle. Dominant species are heather *Calluna vulgaris*, crowberry *Empetrum nigrum* and juniper *Juniperus communis*. The natural westward movement of the bar deposits new ridges for colonisation. Being virtually unaffected by damaging human activities, Culbin Bar is an example of a system with natural structure and function.

Spey Bay

1220 Perennial vegetation of stony banks

Historically, Lower River Spey – Spey Bay in north-east Scotland formed part of the same shingle aggregation as Culbin Bar to the west. Although sea-level rise has separated the sites, they are still linked, being maintained by the same coastal processes. Lower River Spey – Spey Bay and Culbin Bar are, individually, the two largest shingle sites in Scotland and together form a shingle complex unique in Scotland. They represent this habitat type in the northern part of its range in the UK. Lower River Spey – Spey Bay contains significant areas of both bare and naturally vegetated parallel shingle ridges, although some of these have been planted with trees. The most significant feature of the site is the complex of wet and dry vegetation types, depending on the physical relief of the shingle ridges and hollows. Species-rich dry heath and grassland occurs on the ridges, while in the wetter hollows there is species-rich wet heath and transitions to a vegetation type comparable to that of dune slacks. Large areas of scrub, mainly of gorse *Ulex europaeus*, also occur.

Appendix 03 - Shingle Invertebrate lists

This appendix provides a more detailed description of those invertebrates most likely to be associated with shingle habitats in Great Britain. They are grouped under their habitat preferences. RDB - Red Data Book species. Tables 1 and 2 below give an indication of the species represented within the different groups. Information supplied by Roger Morris, English Nature, Peterborough.

Species likely to be associated with shingle or drier habitats on shingle

pRDBK & Endemic *Aphrodes duffieldi* Hemiptera Cicadellidae
A leafhopper, 3.7 - 5mm. long, the male brown with variably developed transverse white bands, the female mottled brownish. Found on grasses, and currently recorded only from Dungeness, Kent. Apparently endemic.

RDB1 *Apostenus fuscus* Araneae Liocranidae
A 3mm long spider known in Britain so far only from coastal shingle at Dungeness, despite the fact it is a woodland spider on the continent.

RDB1 *Pellenes tripunctatus* Araneae Salticidae
A jumping spider found on sparsely vegetated shingle. Only recent records are from Dungeness, East Kent.

RDB1 *Dibolia cynoglossi* Coleoptera Chrysomelidae
A small (2-3mm) flea beetle associated with various species of labiate, including Galeopsis, Mentha, Salvia, Stachys, Ballota and also Cynoglossum. Very rare; scattered old records in southern Britain.

RDB1 *Paroxyna lhommei* Diptera Tephritidae
A picture winged fly which was added to the British list relatively recently (1970s). Originally recorded from Sugarloaf Hill, Folkestone, Kent, it has subsequently been found in the Dover district, at Dungeness and at Sandwich. The host plant is not yet known, although large numbers of adults have been seen on *Senecio* spp., particularly *S.erucifolius* and this may well be the true host. So far the total distribution of this species seems to be confined to the channel coast.

RDB1 *Thalera fimbrialis* Lepidoptera Geometridae Sussex Emerald Moth
Known from shingle beaches in Kent and East Sussex, probably extinct at the latter site. The larva feeds on wild carrot, common ragwort and hoary ragwort, possibly also on yarrow.

pRDB1 *Polyodaspis sulcicollis* Diptera Chloropidae

A small fly whose biology is currently imprecisely known. Members of the family feed on a variety of grasses as stem borers.

pRDB1 *Coleophora galbulipennella* Lepidoptera Coleophoridae
A case bearing micro-moth associated with Nottingham catchfly. In Great Britain this species is only known from Dungeness and Hythe Ranges, East Kent.

pRDB1 *Tinagma balteolella* Lepidoptera Douglasiidae
A micro-moth associated with viper's bugloss. Only known from Kent and East Sussex where it is found in a very few coastal localities.

RDB2 *Trichoncus affinis* Araneae Linyphiidae
A small money spider found among the roots of sparse vegetation on shingle. Recorded from Hants, Sussex, Kent and Suffolk.

RDB2 *Hadena albimacula* Lepidoptera Noctuidae White Spot Moth
A moth found on shingle beaches and chalk or limestone cliffs. Larva on Nottingham catchfly. Very local on the southern coast, Kent, Hampshire and South Devon.

pRDB2 *Xyletinus longitarsis* Coleoptera Anobiidae a wood boring beetle
A small (2.5-4mm long) black wood boring beetle recorded mainly from old oaks but also recorded from dead broom. Formerly widespread in the southern half of England, recent records only from Herefordshire.

pRDB2 *Ethmia bipunctella* Lepidoptera Ethmiidae
Black and white micro-moth. The larva feeds on the flowers and leaves of viper's bugloss, *Symphytum* or other Boraginaceae, pupating in a dead stem or rotten wood. Very local, resident in Kent and East Sussex, occasionally occurs elsewhere on southern and eastern coasts.

pRDB2 *Ethmia terminella* Lepidoptera Ethmiidae
Small black and white moth. The larvae feed on the flowers and unripe seeds of viper's bugloss. Only reliably recorded from Kent and Sussex, on areas of coastal shingle.

pRDB2 *Pediasia fascelinella* Lepidoptera Pyralidae
Frequents sand hills. The larva feeds on various grasses. On the coast of Lincolnshire, Norfolk, Suffolk, Essex and south Devon.

RDB3 *Euophrys browningi* Araneae Salticidae
A jumping spider. Found in tide litter and in empty whelk shells on shingle banks. Recorded from Norfolk, Suffolk, Essex and Kent.

RDB3 *Lathys stigmatisata* Araneae Dictynidae
A small spider so far recorded from only Kent, East Sussex, Cornwall and Lundy Island. Found on coastal heath and shingle.

RDB3 *Phlegra fasciata* Araneae Salticidae
A jumping spider found mainly on sand dunes and sometimes shingle along the south coast of England.

RDB3 *Smicronyx coecus* Coleoptera Curculionidae
A small weevil that feeds on Dodder .

RDB3 *Hylaeus euryscapus* Hymenoptera Colletidae
A yellow-faced bee. Habitats used are soft rock cliffs, landslips, open expanses of sand or shingle and coastal dunes, where a female has been seen entering a burrow in loose sand. Pollen sources are unknown, although visits to a number of flowers have been noted. A rare species recorded from southern coastal counties, with about a dozen post-1970 localities.

RDB3 *Leptothorax tubero-interruptus* Hymenoptera Formicidae
A small ant of warm, sandy or stony areas such as dry heathland or coastal localities. It has been found nesting in (dry?) peat and in moss. Confined to the extreme south of England: reasonably frequent in the New Forest, S. Hants., local on the Dorset heaths; Rye Harbour, E. Sussex and Dungeness, E. Kent.

RDB3 *Myrmica specioides* Hymenoptera Formicidae
A red ant nesting in dry sand in coastal positions. Added to the British list in 1962 from colonies discovered on the sandhills at deal in Kent and has since been found at a small number of other coastal sites, all in Kent. Is probably a native species rather than a recent introduction, as it is easily confused with *M. scabrinodis*.

RDB3 *Calophasia lunula* Lepidoptera Noctuidae Toadflax Brocade Moth
A moth of shingle beaches, waste land and gardens. Larva on yellow toadflax and occasionally other Linaria spp. Resident in Kent and Sussex, also a suspected immigrant.

RDB3 *Eilema pygmaeola* Lepidoptera Arctiidae Pygmy Footman Moth
A moth with two subspecies in Britain. Subsp. pygmaeola; found on coastal sandhills in Kent and Norfolk, immigrant elsewhere. Subsp. *pallifrons*; found on shingle at Dungeness, Kent. The larva is said to feed on unspecified lichens.

RDB3 *Heliothis viriplaca* Lepidoptera Noctuidae Marbled Clover Moth
Breckland, waste places, shingle and sandy beaches and chalk downland, the larva feeding mainly on the flowers of wasteland plants, e.g. *Silene*, *Ononis* etc.

pRDB3 *Apion rubiginosum* Coleoptera Apionidae
A seed weevil of southern distribution, associated with sheep's sorrel (*Rumex acetosella*), apparently with a preference for coastal habitats.

pRDB3 *Cantharis fusca* Coleoptera Cantharidae
A large soldier beetle of wet places. Larvae predators on soil surface. Adults usually on flowers, especially umbellifers. Declining. Now apparently restricted to small part of S Britain where it still can be found in

abundance.

pRDB3 *Ceutorhynchus verrucatus* Coleoptera Curculionidae
A small weevil that appears to be exclusively associated with *Glaucium flavum* in Britain. Adults and larvae occur inside a cavity in the main tap-root. Found on coastal shingle and recorded from Sussex, Essex, Hants, and Devon.

pRDB3 *Dromius vectensis* Coleoptera Carabidae
A small reddish brown ground beetle with conspicuous markings. Lives among vegetation on sandy ground, most often on the coast. Widespread but rare along southern coastal counties.

pRDB3 *Dryophilus anobioides* Coleoptera Anobiidae a wood boring beetle
A small brown beetle, larvae develop in the dead stems of broom, and the adults are usually found by beating dead broom. Occurs in woodlands scrub and roadside verges. Very few modern records.

pRDB3 *Geomyza hendeli* Diptera Opomyzidae
A small fly with a darkened wing tip. The larvae of this family develop in the stems and middle shoots of grasses, though exact host plants of this species are uncertain. The status of this species in Britain is somewhat uncertain. Specimens under this name in the BM(NH) have proved to be *G. apicalis*, but female specimens from Wicken Fen, Cambs and Dungeness, Kent are possibly true *G. hendeli* - males are required for confirmation.

pRDB3 *Acleris permutana* Lepidoptera Tortricidae
Apparently confined to sandhills and other coastal habitats where the larval foodplant *Rosa pimpinellifolia* occurs. Local and scarce, a discontinuous distribution from Kent, Sussex, Cornwall, Derbyshire, Cheshire and parts of Wales.

pRDB3 *Cynaeda dentalis* Lepidoptera Pyralidae
An extremely local micro-moth frequenting coastal localities. The larva feeds in the stem and on the leaf bases of viper's bugloss. Recorded from southern and south-eastern England, from Suffolk to Devon.

pRDB3 *Melissoblaptes zelleri* Lepidoptera Pyralidae
Flies from June to August, larvae feed on *Brachythecium albicans*. A rare species occurring on the coasts of Norfolk,Suffolk and Kent. Also recorded from the Isle of Wight and Glos.

pRDB3 *Platytes alpinella* Lepidoptera Pyralidae
Frequents sandy coasts, the larva feeding on Tortula spp. and other mosses. Very local and rather uncommon, distributed along the south coast from Devon to Kent, also East Anglia, Lincolnshire and Yorkshire.

pRDBK *Astenus procerus* Coleoptera Staphylinidae
Small ground dwelling predatory rove beetle

pRDBK *Hister quadrimaculatus* Coleoptera Histeridae
A carrion beetle

pRDBK *Neofriseria singula* Lepidoptera Gelechiidae a micro-moth
A micro-moth, the larva of which lives in a silken tube on sheep's sorrel *Rumex acetosella*. This species has been much confused with *N. peliella*.

Table 1: Invertebrates listed as Red Data Book species in the Invertebrate Site Register that have relatively strong affinities to shingle habitats

Hemiptera	Aranaea	Coleoptera	Diptera	Lepidoptera	Hymenoptera
Aphrodes duffieldi	*Apostenus fuscus*	*Dibolia cynoglossi*	*Paroxyna lhommei*	*Thalera fimbrialis*	*Hylaeus euryscapus*
	Pellenes tripunctatus	*Xyletinus longitarsis*	*Polyodaspis sulcicollis*	*Coleophora galbulipennella*	*Leptothorax tubero-interruptus*
	Trichoncus affinis	*Smicronyx coecus*	*Geomyza hendeli*	*Tinagma balteolella*	*Myrmica specioides*
	Euophrys browningi	*Apion rubiginosum*		*Hadena albimacula*	
	Lathys stigmatisata	*Cantharis fusca*		*Ethmia bipunctella*	
	Phlegra fasciata	*Ceutorhynchus verrucatus*		*Ethmia terminella*	
		Dromius vectensis		*Pediasia fascelinella*	
		Dryophilus anobioides		*Calophasia lunula*	
		Astenus procerus		*Eilema pygmaeola*	
		Hister quadrimaculatus		*Heliothis viriplaca*	
				Acleris permutana	
				Cynaeda dentalis	
				Melissoblaptes zelleri	
				Platytes alpinella	
				Neofriseria singula	

Green shade = probable dry grassland species.

Species likely to be associated with wetland or damp habitats on shingle structures

RDB1 *Clostera anachoreta* Lepidoptera Notodontidae Scarce hocolate-tip Moth
This species is a relatively recent arrival that has a tenuous foothold at Dungeness, the larva feed on *Salix* and *Populus*.

pRDB2 *Anagnota collini* Diptera Anthomyzidae
An Anthomyzid fly whose larvae develop in the galls of the Chloropid fly *Lipara* formed on *Phragmites*, also recorded from the galls of the mite *Stenotarsonemus*.

pRDB2 *Gelechia muscosella* Lepidoptera Gelechiidae
A micro-moth recorded from fenland and amoungst sallow scrub on coastal shingle. The larva feeds in a catkins of *Poplus* and *Salix* spp.

RDB3 *Hydrophilus piceus* Coleoptera Hydrophilidae Great Silver Water Beetle
Britains largest insect. A large, shiny black water beetle living in weed choked dykes in grazing levels. The adult is phytophagous and detritivorous, but the larvae are predatory on aquatic molluscs. Adults fly readily and are attracted to light at night. Very scarce, confined to the Thames Marshes and other grazing levels in Kent and Sussex, with other populations in South Hampshire and the Somerset and Gwent levels and with a few records in East Anglia. Included in the published Red Data Book as RDB3, but more realistically graded at Notable category A. Distribution map available in Foster (1987).

RDB3 *Minettia flaviventris* Diptera Lauxaniidae a fly

A small fly recorded from herb rich meadows and unimproved pasture, woodland edge and marshes. Larval biology unknown, but other members of the genus have been reared from decomposing vegetable matter such as leaf litter, dead wood and rotting pine cones. Few, widely scattered records throughout Britain.

RDB3 *Hirudo medicinalis* Gnathobdelli Hirudinidae Medicinal Leech
Large leech, formerly widespread and used in phlebotomy. Lives in warm shallow lakes, usually where the temperature exceeds 19oC for much of the summer, including some upland shallow tarns. Feeds on mammals, birds and amphibians, the main host species varying from site to site. Now very rare, still found in a few areas including Cumbria, SE Scotland, Anglesey, East Anglia and the New Forest.
Declined due to over-collecting, changes in land use surrounding suitable lakes and drainage of ponds and lakes. Deepening of ponds for trout fishing, thereby altering their temperature regime, is also a threat. On Appendix II of CITES and schedule 5 of the Wildlife & Countryside Act.

RDB3 *Monosynamma bohemani* Hemiptera Miridae
A rare plantbug. Members of the genus *Monosynamma* are difficult to identify, and records must be treated with care. *M. bohemanni* has so far been certainly identified only from a sandpit in Surrey, where it was associated with creeping willow *Salix repens*. Old records are likely to refer to *M. sabulicola*.

RDB3 *Archanara algae* Lepidoptera Noctuidae Rush Wainscot Moth
Inhabits broadland, freshwater ponds and old water-filled gravel workings. Larva in the stems of *Scirpus lacustris*, *Typha latifolia* and *Iris pseudacorus*. Locally in the Norfolk Broads, also in a few ponds in mid Sussex, noted in a few localities elsewhere in southern England.

RDB3 *Photedes brevilinea* Lepidoptera Noctuidae Fenn's Wainscot
Inhabits fenland and reedy ditches. Larva lives in the stems *Phragmites australis*. Adults fly from mid July to mid August. A very local and rare species confined to Norfolk and Suffolk, it is also rare on the continent.

pRDB3 *Telmatophilus brevicollis* Coleoptera Cryptophagidae
A small brown beetle of southern distribution, found on reedmace and other waterside vegetation.

pRDB3 *Poecilobothrus ducalis* Diptera Dolichopodidae

A small dancefly, larvae probably semi-aquatic carnivore in mud beside saline pools and ditches. Adults recorded from July to September. All modern records are from the North Kent Marshes, though there are also old records from Hants and Somerset.

pRDBK *Ocyusa nigrata* Coleoptera Staphylinidae
Small shiny black rove beetle living among litter in marshes. Scattered localities, mainly in S England. Very rare.

Table 2: Invertebrates listed as RDB in the Invertebrate Site Register that occur in wetlands within shingle structures (not shingle associates)

Leaches	Hemiptera	Coleoptera	Diptera	Lepidoptera
Hirudo medicinalis	*Monosynamma bohemani*	*Hydrophilus piceus*	*Anagnota collini*	*Clostera anachoreta*
		Telmatophilus brevicollis	*Minettia flaviventris*	*Gelechia muscosella*
		Ocyusa nigrata	*Poecilobothrus ducalis*	*Archanara algae*
				Photedes brevilinea

Anthropogenic habitats

RDB1 *Omophron limbatum* Coleoptera Carabidae
Patterned yellowish and bronze green globular ground beetle, living in burrows in bare sandy soil at the margin of water. Very rare, recorded only from Rye Harbour in Sussex and Dungeness in Kent.

RDB1 *Pilophorus confusus* Hemiptera Miridae
There are only two records for this plant bug in Britain, from a Surrey sandpit and from Dungeness. In both cases creeping willow, *Salix repens* was present. The British examples were found largely on the ground, but the species may also occur on trees. The ecology and habitat requirements in Britain are not clearly established.

RDB1 *Nomada ferruginea* Hymenoptera Anthophoridae
A cuckoo bee or cleptoparasite of the mining bee *Andrena praecox*. It has been recorded from a number of counties, mostly in the south-east of England, from Hampshire to Leicestershire and Norfolk. Found breeding on spoil heaps in the Long Pits area of Dungeness.

RDB1 *Hydraecia osseola hucherardi* Lepidoptera Noctuidae Marsh Mallow Moth
The larva feeds on the roots of *Althaea officinalis*. Local, occurring in south-east Kent, East Sussex and also in north Kent. Formerly bred on the Romney Marshes and not a shingle species *per-se*.

pRDB2 *Dyschirius obscurus* Coleoptera Carabidae
Small (3.5-4.5mm) dull black fossorial ground beetle, occurring on bare sand in damp places. Specific predator on rove beetles of the genus *Bledius*, especially *B. arenarius*. Modern records from Kent, Norfolk and Sussex. Very rare but localised colonies can be large.

RDB3 *Heterocerus hispidulus* Coleoptera Heteroceridae
A small black and orange beetle, 3.5mm long. Burrows in soft sand beside gravel pits etc, living in colonies. Only two localities known in Britain, one in Sussex and one in Suffolk.

Vagrants and migrants

RDB2 *Catocala sponsa* Lepidoptera Noctuidae Dark Crimson Underwing
Oak woodlands, the larva on *Quercus*. Now known form the New Forest, Hampshire, formerly known in Wiltshire, Sussex and Kent.

RDB2 *Coscinia cribraria* Lepidoptera Arctiidae Speckled Footman
Very local, restricted to a few heathland areas in Hampshire and Dorset. Larva possibly on low growing herbaceous plants. Subsp.*arenaria* is a rare migrant to Britain.

RDB2 *Scopula nigropunctata* Lepidoptera Geometridae Sub-angled Wave
A small moth.

Extinct: *Nola aerugula* Lepidoptera Nolidae Scarce Black Arches
Formerly restricted on sandhills at Deal, Kent, but since the turn of the century it has been regarded as a migrant in Britain. Continental authors state that the larva feeds on *Lotus* spp.

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Birds of Conservation Concern 4: the population status of birds in the UK, Channel Islands and Isle of Man

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Rosemary Powell

Recent survey data have resulted in the Dotterel *Charadrius morinellus* being one of five upland breeding species that moved from Amber to Red in *BoCC4*.

Abstract This is the fourth review of the status of birds in the UK, Channel Islands and Isle of Man. Using standardised criteria, 244 species were assessed and assigned to the Red, Amber or Green list of conservation concern. The assessment criteria include conservation status at global and European levels and, within the UK, historical decline, trends in population and range, rarity, localised distribution and international importance. The findings are alarming, with 20 species moving on to the Red list and only three leaving it. Three formerly regular breeding species are considered to have ceased breeding in the UK (Temminck's Stint *Calidris temminckii*, Wryneck *Jynx torquilla* and European Serin *Serinus serinus*).

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Some 67 (27.5%) of the UK's regularly occurring bird species are now on the Red list. As well as reinforcing existing conservation concerns, such as for birds of woodland and lowland farmland and for long-distance migrants, this assessment should heighten concern for other groups. Five upland species, including Eurasian Curlew *Numenius arquata* and Dotterel *Charadrius morinellus*, have moved to the Red list. Declines in the UK's internationally important breeding seabird populations are emphasised here by the Red-listing of Shag *Phalacrocorax aristotelis*, Kittiwake *Rissa tridactyla* and Puffin *Fratercula arctica*. Yet the effect of well-targeted conservation action is demonstrated by the recovery of Eurasian Bittern *Botaurus stellaris* and European Nightjar *Caprimulgus europaeus*, with both moving from Red to Amber.

Introduction

This paper presents the fourth 'Birds of Conservation Concern' (*BoCC*) assessment for birds in the UK. Using a well-established approach, based on quantitative assessments against standardised criteria, birds are placed on 'Red', 'Amber' or 'Green' lists to indicate the level of conservation concern we have for them. By using a transparent and standardised approach, based upon the best available data, and conducted by a multi-partner group drawn from relevant organisations in both statutory and non-governmental sectors, this is a robust assessment of the status of all the bird species considered an established part of the UK's avifauna. These lists report on the fortunes of individual species but also indicate broader changes in the UK's biodiversity.

In the last assessment (*BoCC3*, Eaton *et al.* 2009), we stated that 'current pressures on the global environment are unprecedented, with widespread and severe threats to habitats and the species within them', and that funds for conservation action 'are limited, and often the first to be lost in times of economic downturn'. Since then, the pressures on nature on a global scale have increased (Hoekstra & Wiedmann 2014), and the UK has suffered a lengthy and severe economic recession. And, as expected, funding for nature conservation has fallen: public sector spending on biodiversity in the UK has decreased substantially from a recent peak in 2008/09, both in real terms and as a proportion of GDP (Defra 2014). As a consequence of a continuing decline in nature (e.g. Burns *et al.* 2013, Defra 2014), increasing pressures, and decreased resources to tackle these pressures, the need for effective use of those resources has never been greater. The first step to ensure effective use of resources is to prioritise, and exercises such as *BoCC* are essential in this regard, helping us to identify the species (and through further analysis, the countries and regions, habitats, and conservation issues) that most urgently require remedial action.

The red-listing of birds in the UK stretches back over a quarter of a century, with the first formal assessment being that of Batten *et al.* (1990), who listed 117 species in their *Red Data Book*. 'Birds of Conservation Concern' first appeared later that decade, with Gibbons *et al.* (1996b) publishing the first 'traffic light system' of Red, Amber and Green lists. The two subsequent reviews, *BoCC2* (Gregory *et al.* 2002) and *BoCC3* (Eaton *et al.* 2009), have sought to employ the same approach, although there have been some changes in methodology to reflect growing experience and changes in data availability. Key headlines identified during these *BoCC* assessments were:

- *BoCC1* (1996): 36 species were placed on the first *BoCC* Red list, which was instrumental in raising the profile of the severe declines in widespread farmland birds such as Skylark *Alauda arvensis* and Corn Bunting *Emberiza calandra*, part of probably the greatest loss of UK biodiversity in the twentieth century (Aebischer *et al.* 2000).
- *BoCC2* (2002): the Red list rose to 40 species, with the addition of a number of woodland birds such as Lesser Spotted Woodpecker *Dendrocopos minor* and Willow Tit *Poecile montana* illustrating the

bird declines in this habitat (Fuller *et al.* 2005). The continuing recovery of raptors such as Red Kite *Milvus milvus*, Osprey *Pandion haliaetus* and Marsh Harrier *Circus aeruginosus* from historical persecution saw them move from Red to Amber.

- *BoCC3* (2009): a more substantial growth in the Red list saw it expanded to 52 species. The additions to the Red list included more woodland species, such as Hawfinch *Coccothraustes coccothraustes* and Wood Warbler *Phylloscopus sibilatrix*, but for the first time the plight of Afro-Palearctic migrants, such as Common Cuckoo *Cuculus canorus*, rose to the fore, with particular concern for species that winter in the humid tropics (e.g. Vickery *et al.* 2014). Climate change may have contributed to such declines, as it may also have done in the decline of six newly Red-listed northern species (e.g. Whimbrel *Numenius phaeopus* and Redwing *Turdus iliacus*), for which the UK lies at the southern or western edge of the breeding range. Some comfort could be gained from the fact that targeted conservation action for Stone-curlew *Burhinus oedicnemus* and Woodlark *Lullula arborea* resulted in those two species moving from Red to Amber. Notably, *BoCC3* conducted the first subspecies-level *BoCC* assessment, enabling different levels of concern to be applied to different races of the same species (such as Black-tailed Godwits *Limosa limosa* of the nominate race and *L. l. islandica*), and the threats facing some of the UK's endemic races to be identified.

This latest report comes six years after *BoCC3*. The six-year gap fits into an emerging cycle of reporting on the status of the UK's birds, influenced by the requirements of the EU's Wild Birds Directive (79/409/EEC). This dictates that all EU Member States report on the status (e.g. trends, ranges and populations) of all native bird species at six-year intervals. This was most recently done in 2013. The collation of similar data from across the EU, combined with parallel submissions from BirdLife International partners in non-EU countries, enables Europe-wide reporting (European Commission 2015) and the production of an updated European Red List of Birds (ERLOB; BirdLife International 2015) at regular intervals. Through this schedule, a number of the 'building blocks' of *BoCC* assessments are put in place: updated Global and European Red List assessments, and new population estimates through the work of the Avian Population Estimates Panel (APEP; see Musgrove *et al.* 2013), which help us to assess species against criteria for rarity and international importance (see below), the latter helped by the availability of the European dataset compiled for ERLOB.



Edmund Fellowes/BTO

416. The Grey Wagtail *Motacilla cinerea* is one of five upland species moving from Amber to Red in *BoCC4*, a move that highlights concern about species of our upland streams and rivers.

Methods

The species list

As in previous assessments, we considered all naturally occurring native species on the British List (BOU 2013; see also www.bou.org.uk), but with filters to exclude some species from the full assessment: vagrants, defined as species considered by BBRC (www.bbrc.org.uk), and species occurring only as scarce migrants (e.g. White & Kehoe 2015a,b). As before, we have also included Globally Threatened species (BirdLife International 2015) that have occurred in the UK in each of the last 25 years (Balearic Shearwater *Puffinus mauretanicus* and Aquatic Warbler *Acrocephalus paludicola*), regardless of scarcity in the UK.

A number of non-native species are well established in the UK but, despite the fact that some are appreciated by birdwatchers and the public, we do not consider these species to have conservation value in the UK and they are excluded from this assessment.

As in *BoCC3*, rarer breeding species were considered only if they had been proven (or strongly suspected) to breed for five consecutive years within the most recent 25 years for which data are available. This excluded a number of species, such as European Bee-eater *Merops apiaster*, which remain occasional breeders in the UK, and others that may well be in the process of establishing (e.g. Great White Egret *Ardea alba* and Little Bittern *Ixobrychus minutus*). Species considered to be regular breeders in *BoCC3* were excluded from consideration (and placed on the list of 'former breeders') if they had not bred in any of the five most recent years for which data are available.

Note that some species were excluded from assessment as breeding species, but were assessed because they have larger or better-established non-breeding populations (e.g. Red-necked Grebe *Podiceps grisegena* and Black Tern *Chlidonias niger*).

One species was added to our list: Caspian Gull *Larus cachinnans* was assessed for the first time since its acceptance as a full species in 2007 (BOU 2008). Since the last review it has become apparent that the Caspian Gull is a regular non-breeding visitor to the UK.

The assessment process

BoCC assessments use a set of quantitative criteria that fall into two groups, for the Red and the Amber lists. All species are assessed against all of these criteria, and are placed on the highest priority list for which they qualify. If they meet none of these criteria, they are placed on the Green list.

The criteria used for *BoCC4* were largely those used for *BoCC3*, which in turn had evolved from previous *BoCC* assessments. The clear advantage to maintaining a consistent approach to assessments over time is that it allows a direct comparison of the results of those assessments. A few minor adjustments were necessary, to allow for changing circumstances and data availability, and these are outlined below. All the *BoCC* criteria are summarised briefly, but Eaton *et al.* (2009) contained further details, while a fuller account of the criteria and data used is available in the Supplementary Online Material at www.britishbirds.co.uk/wp-content/uploads/2014/07/SM.pdf The adjustments arose because we felt that the criteria used for assessing recovery (and any lapse in that recovery) from historical decline could be improved; because of changes forced upon us by the availability of information on European status; and because of the availability of new atlas data for assessing non-breeding range change. Our adjustments and the reasoning behind them are discussed below, and the impacts of these changes are analysed in the Results section.

Red-list criteria

IUCN: Global conservation status. Species that are Globally Threatened (Critically Endangered, Endangered and Vulnerable, but not Near Threatened) under IUCN guidelines, as assessed by BirdLife International, the IUCN Red List Authority for birds, in 2015 (www.iucnredlist.org).

HD: Historical decline in breeding populations. Species judged to have declined severely between 1800 and 1995, from an assessment conducted by Gibbons *et al.* (1996a), and which have not recovered subsequently. The process by which species should be deemed to have shown partial recovery from historical decline (hence move



Richard Chandler

417. One of the headline birds of this current *BoCC* review is the Eurasian Curlew *Numenius arquata*, which moved from Amber to Red. A recent paper in *BB* called this species the most important bird conservation priority in the UK (Brown *et al.* 2015).

to the Amber list), or complete recovery (move to the Green list), or subsequently faltered from those recoveries, was a subject of much debate. We agreed that the initial assessments of historical decline by Gibbons *et al.*, based on a semi-quantitative scoring of population changes within five periods, were robust; and that it was still appropriate that any HD species doubling its population size or more within the relevant 25-year period, and exceeding 100 breeding pairs, should move to the Amber list (provided it did not qualify as Red under other criteria). We made one change to this step to be consistent with other criteria, and introduced an assessment of trend over the longer-term period, defined as the entire period used for assessments since the first *BoCC* review, starting in 1969.

A key concern, however, was how to treat changes subsequent to a move to Amber (HDrec), namely how any future recovery or decline should be regarded. The criterion used for *BoCC3* stipulated that a decline of 20% between *BoCC* reviews should dictate that a species returns to the Red list, whereas a further increase of 20% over a similar period would enable a species to move to the Green list (unless it qualified as Amber under any other criteria). We felt that this was a rather unsatisfactory approach, in that in both cases the criterion used a non-standard measurement period unrelated to those used for other *BoCC* criteria, and which could lead to changes in status due to relatively insubstantial and short-term fluctuations in population size.

Therefore, for *BoCC4* we have used the following rationale: a species should be moved to the Green list (if not qualifying against other Red or Amber criteria) if it shows continued and substantial recovery from historical decline beyond the level (HDrec) that qualified the species for the Amber list. When it moves to Green, the species should be considered as having recovered permanently and would no longer be considered against the historical decline criterion, i.e. any subsequent decline would be assessed only against the relevant decline criteria such as BDp (see below). That being the case, we felt that at least another doubling of numbers should be required to permit movement to the Green list. In fact, we now require a species to have shown a further

increase of at least 167% from its HDrec level in order to move to the Green list. This higher threshold ensures that if a species subsequently declines by anything less than 25% (thus does not trigger a return to the Amber list under the moderate decline criterion), it will still remain at more than double its HDrec numbers.

As an example, imagine a hypothetical species that qualified for the *BoCC1* Red list under the historical decline criterion, but no others. This species increased from 100 to 300 pairs within 25 years (well over the doubling to 200 required) and thus was moved from Red to Amber in *BoCC2*. If, by the time of this current review, it had increased to 900 pairs (an increase of 200% from its HDrec level of 300 pairs and thus above the 167% threshold of 801 pairs), it would be moved to the Green list and the HD criterion would no longer apply. If it had failed to increase by this rate, but remained above 200 pairs, it would stay on the Amber list. Finally, if it had declined to below 200 pairs, it would return to the Red list. In the last two cases, the HD criterion would still play a role in future assessments.

BDp: Breeding population decline. Severe decline in the UK breeding population size (>50%) over 25 years (**BDp¹**) or the longer-term (**BDp²**), defined as the entire period used for assessments since the first *BoCC* review, starting in 1969.

WDp: Non-breeding population decline. Severe decline in the UK non-breeding population size (>50%) over 25 years (**WDp¹**) or the longer-term (**WDp²**) as defined above. Non-breeding trends were assessed only if a species has substantially independent breeding and non-breeding populations, otherwise only the breeding population was assessed. The same was true for other criteria which could be applied to both breeding and non-breeding populations.

BDr: Breeding range decline. Severe decline in UK range (>50%) between the breeding bird atlases in 1988–91 and 2007–11 (**BDr¹**) or 1968–71 and 2007–11 (**BDr²**), as measured by the calculated change in the number of occupied 10-km squares.

WDr: Non-breeding range decline. Severe decline in UK range (>50%) between the wintering bird atlases in 1981–84 and 2007–11 (**WDr¹**), as measured by the calculated change in the number of occupied 10-km squares. Since there are only two wintering bird atlases, it was not possible to measure range change over a longer time period. Note that while *BoCC* reviews have always intended to assess range change in the non-breeding season, this is the first assessment able to do so.

Amber-list criteria

ERLOB: European Red List status. Previous *BoCC* assessments have used Species of European Conservation Concern assessments (SPECs; see Tucker & Heath 1994 and BirdLife International 2004) as an indication of wider regional concern for a species, and thus Amber-listed any UK species that was SPEC-listed. Although a new assessment of species status across Europe, the European Red List of Birds (ERLOB; BirdLife International 2015), was published in 2015, this produced only IUCN Red List assessments of regional extinction risk (IUCN 2012) with no consideration of the wider suite of measures (species rarity, localisation, moderate decline and depletion) included in SPEC assessments. At present, it is not clear when or if new SPECs will be published. Therefore, to complete the *BoCC* assessment, we faced a quandary: to delay publication of *BoCC* in the hope that SPEC assessments would be completed or to drop the use of SPECs as part of *BoCC*. We chose the latter option, and thus have Amber-listed any species on the European Red List (Critically Endangered, Endangered or Vulnerable). We recognise that the exclusion of species that were previously SPEC-listed has had an impact on our final lists, by moving species from Amber to Green – and we investigate the scale of this impact below – but we feel that our decision provides a sound basis for this and future *BoCC* assessments.

HDrec: Historical decline – recovery. As described above, previously Red-listed for historical decline, followed by an increase of at least 100% over 25 years or the longer-term period. This also applies if the move to

HDrec happened in a previous *BoCC* assessment, having remained above the 100% increase threshold, but not having recovered further to move to Green (see text under historical decline above).

BDMp: Breeding population decline. As for Red-list criterion BDp, but with moderate decline (>25% but <50%) over 25 years (**BDMp¹**) or the longer-term period (**BDMp²**).

WDMp: Non-breeding population decline. As for Red-list criterion WDp, but with moderate decline (>25% but <50%) over 25 years (**WDMp¹**) or the longer-term period (**WDMp²**).

BDMr: Breeding range decline. As for Red-list criterion BDr, but with moderate decline (>25% but <50%) between 1988–91 and 2007–11 (**BDMr¹**) or 1968–71 and 2007–11 (**BDMr²**).

WDMr: Non-breeding range decline. As for Red-list criterion WDr, but with moderate decline (>25% but <50%) between 1981–84 and 2007–11 (**WDMr¹**).

BR & WR: Breeding and non-breeding rarity. Species qualified as rare breeders (BR) if the UK breeding population was <300 pairs, and as rare non-breeders (WR) if the UK non-breeding population was <900 individuals.

BL & WL: Breeding and non-breeding localisation. Species were considered localised if more than 50% of the UK population was found at ten or fewer sites in either the breeding (BL) or the non-breeding (WL) season. Sites were defined as either Special Protection Areas (SPAs; Stroud *et al.* 2001) or Important Bird Areas (IBAs; Heath & Evans 2000). Rare breeders or rare non-breeders (see above) were not assessed against this criterion, as their small population sizes predispose them to be restricted to a small number of sites.

BI & WI: Breeding and non-breeding international importance. Species were considered of international importance if the UK holds at least 20% of the European population in either the breeding (BI) or the non-breeding (WI) season. European estimates were derived from data collated as part of the ERLOB assessment, but for non-breeding waterbirds we used estimates for the flyway populations for northwest Europe (wildfowl) or East Atlantic (waders) (Wetlands International 2015).

Data sources

We are fortunate in that, thanks to the efforts of thousands of dedicated volunteer birdwatchers working in tandem with professional research and conservation organisations, birds in the UK are one of the best-monitored taxonomic groups anywhere in the world. We are thus well equipped to make status assessments such as *BoCC*, and for many species can make robust assessments against all the *BoCC* criteria. This is not true for all species, however, and it is highly likely that some data gaps have influenced our assessment. The principal sources of data were as for *BoCC3*, and our treatment of data from these sources was as described in Eaton *et al.* (2009). Further details can be found at www.britishbirds.co.uk/wp-content/uploads/2014/07/SM.pdf

In summary, the main sources for measuring population trends were:

- The *BTO/JNCC Common Birds Census (CBC)* and *BTO/JNCC/RSPB Breeding Bird Survey (BBS)*; when combined, these gave us trends for common and widespread breeding birds from the late 1960s onwards. For some species, such as Common Swift *Apus apus* and Wood Warbler, trends were available only from the start of the BBS in 1994. Details of the BBS and the latest results can be found in Harris *et al.* (2015) and at www.bto.org/volunteer-surveys/bbs
- *BTO/JNCC/RSPB Wetland Bird Survey (WeBS)* and *WWT/JNCC/SNH Goose and Swan Monitoring Programme*, which together provided annual trends for most wildfowl species from 1966/67 onwards and for waders from 1974/75 onwards, with a few other waterbird species monitored over shorter periods. See Holt *et al.* (2015) and www.bto.org/volunteer-surveys/webs and http://monitoring.wwt.org.uk/our-work/goose-swan-monitoring-programme

- Seabird monitoring comes from two sources: the three complete censuses conducted in 1969–70 (Cramp *et al.* 1974), 1985–88 (Lloyd *et al.* 1991) and 1998–2001 (Mitchell *et al.* 2004), and the Seabird Monitoring Programme that has monitored a UK-wide sample of colonies since 1986. See www.jncc.defra.gov.uk/page-1550
- Rare Breeding Birds Panel data provided trends since 1973 for rare breeders (defined, loosely, as species with UK populations of less than 2,000 pairs, although data collation for less rare species began more recently than 1973). We used data up to 2012 (Holling *et al.* 2014) to create long-term and 25-year trends, sometimes in combination with estimates from single-species surveys. See www.rbbp.org.uk
- Periodic species surveys run under the Statutory Conservation Agency and RSPB Annual Breeding Birds Scheme (SCARABBS) programme, BTO species surveys and the GWCT/BTO Woodcock survey provided trends and population figures for a number of scarce and rare species.

With occasional exceptions (see the Supplementary Online Material for details), trends were calculated using data up to and including 2012. In the case of BBS/CBC and BBS trends, these were smoothed trends, using data from 2013 but changes reported up to 2012 following standard statistical practice.

For measuring trends in range we relied on the three breeding bird atlases (Sharrock 1976, Gibbons *et al.* 1993 and Balmer *et al.* 2013) and two wintering bird atlases (Lack 1986 and Balmer *et al.* 2013). Given the 20-year gaps between breeding atlases, some *BoCC* assessments (e.g. *BoCC3*) have been forced to rely on rather out-of-date measures of change in range. The recent *Bird Atlas 2007–11* allowed us to generate up-to-date measures of change in breeding range over both the long-term (between the first and third atlases, a period of 40 years) and a 20-year period (between the second and third atlases, approximating to the 25-year trend period). In addition, we were able for the first time to calculate (near) 25-year trends in non-breeding range, based on the two winter atlases with fieldwork periods covering 1981/82 to 1984/85 and 2007/08 to 2010/11.



Andy Hay/RSPB-images

418. Three breeding seabirds moved from Amber to Red in *BoCC4*, with both Shag *Phalacrocorax aristotelis* (illustrated) and Kittiwake *Rissa tridactyla* doing so because of continuing serious declines in the UK breeding populations.

Population estimates were derived from a range of sources and almost all are as reported by APEP (see Musgrove *et al.* 2013). To maintain consistency with the data used for UK reporting under the Wild Birds Directive, we did not update these estimates to account for any additional data available since their publication, except for species for which the results from new national surveys were available (e.g. Dotterel *Charadrius morinellus*; Hayhow *et al.* 2015). Localisation estimates were derived using these UK estimates and data collated in the third review of the UK's network of SPAs (Stroud *et al.* in prep.). There has been no update of the population estimates within IBAs since the *BoCC3* review; since these form an important complementary approach to assessing localisation within SPAs, we simply reused the existing *BoCC3* assessments for IBAs.

Race-level assessments

As with *BoCC3*, we conducted a parallel assessment of the *BoCC* status of regularly occurring races of birds. With the exception of the changes in criteria (HD and ERLOB) described above and applied similarly to races, the process was as described in Eaton *et al.* (2009). As before, the lack of some data sources at a race level (e.g. Global and European IUCN assessments, and monitoring data at the race level) required us to create new estimates of populations, trends and status outside of the UK as best we could with existing data sources.

We note that over the last six years little has changed to clarify further the status of some of the UK's less well-known races. We used as our starting point the same list of races compiled for *BoCC3*, based primarily on the list of races maintained by the BOU but informed by other key references; as before, our inclusion of a race in this review does not constitute a judgement on its validity. There were, however, some relevant taxonomic changes, most arising from investigations at the species level leading to changes in what is considered a valid race. Four races were no longer considered: Pintail *Anas acuta* and Sandwich Tern *Sterna sandvicensis* are now considered monotypic following the split of other races into separate species (Southern Pintail *A. eatoni* and Cabot's Tern *S. acuflavida*, respectively), Red Kite is effectively monotypic following the extinction of the Cape Verde Kite *M. m. fasciicauda* (Johnson *et al.* 2005), and the occurrence of the Marsh Tit race *Poecile p. palustris* in the UK was dismissed by Broughton (2009). We considered three additional races: Greater Scaup *Aythya m. marila*, European Storm-petrel *Hydrobates p. pelagicus* and Slavonian Grebe *Podiceps a. auritus*, as a consequence of these species being recognised as polytypic since our last assessment (del Hoyo & Collar 2014).

Results

BoCC4 species-level assessment

Three species were identified as not having bred in the UK in the last five years for which data were available; they were thus removed from the assessment and are now considered to be 'former breeders': Temminck's Stint *Calidris temminckii*, Wryneck *Jynx torquilla* and European Serin *Serinus serinus* (table 1). The addition of Caspian Gull meant that in total 244 species were assessed. Of these 244 species, *BoCC4* placed 67 (27.5%) on the Red list, 96 (39.3%) on the Amber list, and 81 (33.2%) on the Green list. Lists of species, qualifying criteria and values are given in tables 2–4.

There has been a substantial change in the way species are distributed among the three

Table 1. Formerly regular breeding species in the UK.

	year of last recorded breeding
Great Bustard *Otis tarda*	c. 1833
Kentish Plover *Charadrius alexandrinus*	1979
Temminck's Stint *Calidris temminckii*	1993
Black Tern *Chlidonias niger*	1975
Great Auk *Pinguinus impennis*	c. 1812
Snowy Owl *Bubo scandiacus*	1975
Wryneck *Jynx torquilla*	2002
European Serin *Serinus serinus*	2006

Table 2. Species on the *BoCC4* Red list, the criteria under which they qualify, and values for those criteria. Red[b] and Amber[c] criteria

	BoCC3[a]	IUCN	HD	BDp[1] & BDMp[1]	BDp[2] & BDMp[2]	WDp[1] & WDMp[1]	WDp[2] & WDMp[2]	BDr[1] & BDMr[1]	BDr[2] & BDMr[2]	WDr[1] & WDMr[1]	ERLOB	HDrec	BR	WR	BL	WL	BI	WI
White-fronted Goose *Anser albifrons*	G					-60	-54											
Common Pochard *Aythya ferina*	A	VU				-51	-34	-38	-36		VU							
Greater Scaup *Aythya marila*	R					-49	-83				VU					90–100[IBA]		
Long-tailed Duck *Clangula hyemalis*	G	VU									VU							
Common Scoter *Melanitta nigra*	R			-63	-81			-58	-50				52			90–100[Both]		
Velvet Scoter *Melanitta fusca*	A	VU									VU					90–100[IBA]		
Black Grouse *Tetrao tetrix*	R		*	-80	Severe decline				-28									
Capercaillie *Tetrao urogallus*	R			-42	Severe decline				-72	-55								
Grey Partridge *Perdix perdix*	R			-76	-91				-40									
Balearic Shearwater *Puffinus mauretanicus*	R	CR									CR							
Shag *Phalacrocorax aristotelis*	A			-62	-36												50–60	
Red-necked Grebe *Podiceps grisegena*	A					-72								57				
Slavonian Grebe *Podiceps auritus*	A	VU		-56	-62			-33					30					
White-tailed Eagle *Haliaeetus albicilla*	R		*										37–44					
Hen Harrier *Circus cyaneus*	R		*															
Corn Crake *Crex crex*	R				-52				-74			*						
Northern Lapwing *Vanellus vanellus*	R			-57	-63						VU							
Ringed Plover *Charadrius hiaticula*	A			-37		-52	-42											50–60
Dotterel *Charadrius morinellus*	A			-57											70–80[Both]			
Whimbrel *Numenius phaeopus*	R			-67				-50	-27									
Eurasian Curlew *Numenius arquata*	A			-49	-62						VU						20–30	
Black-tailed Godwit *Limosa limosa*	R		*					-35			VU		61–66			90–100[IBA]		
Ruff *Calidris pugnax*	R			-41	-62			-62					0–11	820				
Red-necked Phalarope *Phalaropus lobatus*	R		*										20–24					
Woodcock *Scolopax rusticola*	A			-29				-31	-52									
Arctic Skua *Stercorarius parasiticus*	R			-79	-44													
Puffin *Fratercula arctica*	A	VU							-27		EN				80–90[Both]			
Roseate Tern *Sterna dougallii*	R			-73	-86			-27	-62				89					

Table 2. Species on the *BoCC4* Red list, the criteria under which they qualify, and values for those criteria. Red[b] and Amber[c] criteria (cont.)

	BoCC3[a]	IUCN	HD	BDp[1] & BDMp[1]	BDp[2] & BDMp[2]	WDp[1] & WDMp[1]	WDp[2] & WDMp[2]	BDr[1] & BDMr[1]	BDr[2] & BDMr[2]	WDr[1] & WDMr[1]	ERLOB	HDrec	BR	WR	BL	WL	BI	WI
Kittiwake *Rissa tridactyla*	A			-74	-62						VU							
Herring Gull *Larus argentatus*	R				-60	-53 to -60												20–30
Turtle Dove *Streptopelia turtur*	R	VU		-92	-96			-35	-51		VU							
Common Cuckoo *Cuculus canorus*	R			-60	-62													
Lesser Spotted Woodpecker *Dendrocopos minor*	R			-81	-81			-29	-37									
Merlin *Falco columbarius*	A		*															
Golden Oriole *Oriolus oriolus*	R			-81	-70			-40					2–5					
Red-backed Shrike *Lanius collurio*	R		*	-66	-93				-88				1–3					
Willow Tit *Poecile montana*	R			-91	-94			-49	-54	-43								
Marsh Tit *Poecile palustris*	R			-43	-72													
Skylark *Alauda arvensis*	R			-32	-62													
Wood Warbler *Phylloscopus sibilatrix*	R			-66				-37	-34									
Grasshopper Warbler *Locustella naevia*	R			-68	-93													
Savi's Warbler *Locustella luscinioides*	R			-74	-66			-30					1–3					
Aquatic Warbler *Acrocephalus paludicola*	R	VU									VU							
Marsh Warbler *Acrocephalus palustris*	R			-77	-88								2–8					
Common Starling *Sturnus vulgaris*	R			-70	-83													
Ring Ouzel *Turdus torquatus*	R			-72					-43									
Fieldfare *Turdus pilaris*	R			-50	-63			-77	-32				1–2					
Song Thrush *Turdus philomelos*	R				-59													
Redwing *Turdus iliacus*	R			-73	-49			-45	-32				4–16					
Mistle Thrush *Turdus viscivorus*	A			-45	-62													
Spotted Flycatcher *Muscicapa striata*	R			-80	-88													
Common Nightingale *Luscinia megarhynchos*	A			-60	-85				-43									
Pied Flycatcher *Ficedula hypoleuca*	A			-53				-27										
Black Redstart *Phoenicurus ochruros*	A			-52	-33								19–44	400				
Whinchat *Saxicola rubetra*	A			-55				-38	-48									

Table 2. Species on the *BoCC4* Red list, the criteria under which they qualify, and values for those criteria. Red[b] and Amber[c] criteria (cont.)

	BoCC3[a]	IUCN	HD	BDp¹ & BDMp¹	BDp² & BDMp²	WDp¹ & WDMp¹	WDp² & WDMp²	BDr¹ & BDMr¹	BDr² & BDMr²	WDr¹ & WDMr¹	ERLOB	HDrec	BR	WR	BL	WL	BI	WI
House Sparrow *Passer domesticus*	R			-32	-66													
Tree Sparrow *Passer montanus*	R				-90				-37									
Yellow Wagtail *Motacilla flava*	R			-63	-70			-25	-32									
Grey Wagtail *Motacilla cinerea*	A			-33	-57													
Tree Pipit *Anthus trivialis*	R			-64	-70				-29									
Hawfinch *Coccothraustes coccothraustes*	R			-74				-64	-75									
Linnet *Linaria cannabina*	R				-60													
Twite *Linaria flavirostris*	R		*	-52														
Lesser Redpoll *Acanthis cabaret*	R			-64	-83												30–40	
Yellowhammer *Emberiza citrinella*	R			-49	-54													
Cirl Bunting *Emberiza cirlus*	R								-85									
Corn Bunting *Emberiza calandra*	R		*	-61	-90			-35	-56	-27								

[a] *BoCC3* assessments: R = Red, A = Amber, G = Green

[b] Red-list criteria:

IUCN: Globally Threatened (CR = Critically Endangered, EN = Endangered, VU = Vulnerable). HD: historical decline in the breeding population. BDp¹/²: severe breeding population decline over 25 years/longer term. WDp¹/²: severe non-breeding population decline over 25 years/longer term. BDr¹/²: severe breeding range decline over 25 years/longer term. WDr¹: severe non-breeding range decline over 25 years.

[c] Amber-list criteria:

ERLOB: Threatened in Europe (CR = Critically Endangered, EN = Endangered, VU = Vulnerable). HDrec: historical decline – recovery. BDMp¹/²: moderate breeding population decline over 25 years/longer term. WDMp¹/²: moderate non-breeding population decline over 25 years/longer term. BDMr¹/²: moderate breeding range decline over 25 years/longer term. WDMr¹: moderate non-breeding range decline over 25 years. BR/WR: breeding/non-breeding rarity. BL/WL: breeding/non-breeding localisation. Superscript text indicates whether species qualified as localised in IBAs, SPAs, or both. BI/WI: breeding/non-breeding international importance. Figures are given in bands for species exceeding the qualifying thresholds for the localisation and international importance criteria.

Red and Amber criteria for population and range trends are given as % change, and are combined in the same columns (e.g. BDp¹ and BDMp¹): red and amber colour-coding is used to show which a species qualified against.

When a species has changed list since *BoCC3*, shading of table cells is used to indicate the criteria responsible for that change.

Table 3. Species on the *BoCC4* Amber list, the criteria under which they qualify, and values for those criteria. Amber criteria[b]

	BoCC3[a]	ERLOB	HDrec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	BDMr[2]	WDMr[1]	BR	WR	BL	WL	BI	WI
Mute Swan *Cygnus olor*	G															20–30
Bewick's Swan *Cygnus columbianus*	A	EN				-45								90–100[Both]		30–40
Whooper Swan *Cygnus cygnus*	A										9–14			90–100[IBA]		
Bean Goose *Anser fabalis*	A											730				
Pink-footed Goose *Anser brachyrhynchus*	A													90–100[Both]		70–80
Greylag Goose *Anser anser*	A													50–60[IBA]		
Barnacle Goose *Branta leucopsis*	A													70–80[Both]		
Brent Goose *Branta bernicla*	A													70–80[Both]		40–50
Common Shelduck *Tadorna tadorna*	A			-41										50–60[IBA]	20–30	20–30
Eurasian Wigeon *Anas penelope*	A													50–60[IBA]		30–40
Gadwall *Anas strepera*	A															20–30
Eurasian Teal *Anas crecca*	A															40–50
Mallard *Anas platyrhynchos*	A					-38										
Pintail *Anas acuta*	A					-34		-27	-27		9–33			90–100[IBA]		40–50
Garganey *Anas querquedula*	A										14–93					
Shoveler *Anas clypeata*	A															30–40
Common Eider *Somateria mollissima*	A	VU														
Common Goldeneye *Bucephala clangula*	A										200					
Smew *Mergellus albellus*	A											180				
Common Quail *Coturnix coturnix*	A		*													
Red Grouse *Lagopus lagopus*	A	VU														
Black-throated Diver *Gavia arctica*	A										220	560				
Great Northern Diver *Gavia immer*	A	VU														50–60
Fulmar *Fulmarus glacialis*	A	EN											50–60[IBA]			
Manx Shearwater *Puffinus puffinus*	A								-28				90–100[Both]		80–90	
European Storm-petrel *Hydrobates pelagicus*	A												90–100[Both]			
Leach's Storm-petrel *Oceanodroma leucorhoa*	A												90–100[Both]			
Northern Gannet *Morus bassanus*	A												90–100[Both]		30–40	

Table 3. Species on the *BoCC4* Amber list, the criteria under which they qualify, and values for those criteria. Amber criteria[b] (cont.)

	BoCC3[a]	ERLOB	HD rec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	BDMr[2]	WDMr[1]	BR	WR	BL	WL	BI	WI
Eurasian Bittern *Botaurus stellaris*	R		*								80	600				
Eurasian Spoonbill *Platalea leucorodia*	A										2	20				
Black-necked Grebe *Podiceps nigricollis*	A										32–51	130				
Honey-buzzard *Pernis apivorus*	A										33–69					
Marsh Harrier *Circus aeruginosus*	A		*										50–60[SPA]			
Montagu's Harrier *Circus pygargus*	A								-28		12–16					
Osprey *Pandion haliaetus*	A		*								200–250					
Spotted Crake *Porzana porzana*	A										27					
Common Crane *Grus grus*	A										9–14	52				
Stone-curlew *Burhinus oedicnemus*	A								-42				70–80[Both]			
Avocet *Recurvirostra avosetta*	A												90–100[Both]/90–100[Both]			
Oystercatcher *Haematopus ostralegus*	A	VU												50–60[Both]	30–40	30–40
Grey Plover *Pluvialis squatarola*	A													60–70[Both]		20–30
Bar-tailed Godwit *Limosa lapponica*	A													80–90[Both]		30–40
Turnstone *Arenaria interpres*	A					-41										20–30
Red Knot *Calidris canutus*	A													90–100[Both]		40–50
Curlew Sandpiper *Calidris ferruginea*	G	VU														
Sanderling *Calidris alba*	G													60–70[SPA]		
Dunlin *Calidris alpina*	R						-49	-27					70–80[Both]/50–60[SPA]			
Purple Sandpiper *Calidris maritima*	A			-43		-49		-33			1					
Common Sandpiper *Actitis hypoleucos*	A			-45	-40											
Green Sandpiper *Tringa ochropus*	A										1–3					
Spotted Redshank *Tringa erythropus*	A											98				
Greenshank *Tringa nebularia*	G												70–80[SPA]			
Wood Sandpiper *Tringa glareola*	A										11–27	<900				
Common Redshank *Tringa totanus*	A			-44		-32		-35	-43							20–30
Common Snipe *Gallinago gallinago*	A								-31							
Great Skua *Stercorarius skua*	A												70–80[SPA]		50–60	

Table 3. Species on the *BoCC4* Amber list, the criteria under which they qualify, and values for those criteria. Amber criteria[b] (cont.)

	BoCC3[a]	ERLOB	HDrec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	BDMr[2]	WDMr[1]	BR	WR	BL	WL	BI	WI
Black Guillemot *Cepphus grylle*	A							-29								
Razorbill *Alca torda*	A												70–80[Both]		20–30	
Common Guillemot *Uria aalge*	A												50–60[IBA]		50–60	
Little Tern *Sternula albifrons*	A								-30				60–70[Both]			
Sandwich Tern *Sterna sandvicensis*	A			-25									90–100[Both]			
Common Tern *Sterna hirundo*	A												60–70[IBA]			
Arctic Tern *Sterna paradisaea*	A			-38				-29								
Black-headed Gull *Chroicocephalus ridibundus*	A					-33 to -41										60–70
Mediterranean Gull *Larus melanocephalus*	A												50–60[IBA]			
Common Gull *Larus canus*	A															40–50
Caspian Gull *Larus cachinnans*	NA											90				
Lesser Black-backed Gull *Larus fuscus*	A												70–80[IBA]		20–30	
Glaucous Gull *Larus hyperboreus*	A											170				
Iceland Gull *Larus glaucoides*	A											240				
Yellow-legged Gull *Larus michahellis*	A										1					
Great Black-backed Gull *Larus marinus*	A				-29	-33 to -58										
Stock Dove *Columba oenas*	A														20–30	
Tawny Owl *Strix aluco*	G			-31	-30											
Short-eared Owl *Asio flammeus*	A							-38	-47							
European Nightjar *Caprimulgus europaeus*	R								-45							
Common Swift *Apus apus*	A			-38												
Common Kingfisher *Alcedo atthis*	A	VU														
Common Kestrel *Falco tinnunculus*	A			-33	-46											
Shore Lark *Eremophila alpestris*	A											74				
House Martin *Delichon urbicum*	A			-33	-49											
Willow Warbler *Phylloscopus trochilus*	A			-32	-38											
Dartford Warbler *Sylvia undata*	A		*										50–60[SPA]			
Short-toed Treecreeper *Certhia brachydactyla*	A										<300					

Table 3. Species on the *BoCC4* Amber list, the criteria under which they qualify, and values for those criteria. Amber criteria[b] (cont.)

	BoCC3[a]	ERLOB	HDrec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	BDMr[2]	WDMr[1]	BR	WR	BL	WL	BI	WI
Dipper *Cinclus cinclus*	G			-27												
Common Redstart *Phoenicurus phoenicurus*	A								-31							
Dunnock *Prunella modularis*	A				-31											
Meadow Pipit *Anthus pratensis*	A				-44											
Water Pipit *Anthus spinoletta*	A											190				
Bullfinch *Pyrrhula pyrrhula*	A				-39											
Common Redpoll *Acanthis flammea*	G											310				
Scottish Crossbill *Loxia scotica*	A														100	
Parrot Crossbill *Loxia pytyopsittacus*	A										65					
Snow Bunting *Plectrophenax nivalis*	A										100					
Lapland Bunting *Calcarius lapponicus*	A											710				
Reed Bunting *Emberiza schoeniclus*	A				-38											

[a] *BoCC3* assessments:

NA = Not assessed, R = Red, A = Amber, G = Green

[b] Amber-list criteria:

ERLOB: Threatened in Europe (CR = Critically Endangered, EN = Endangered, VU = Vulnerable). HDrec: historical decline – recovery. BDMp[1/2]: moderate breeding population decline over 25 years/longer term. WDMp[1/2]: moderate non-breeding population decline over 25 years/longer term. BDMr[1/2]: moderate breeding range decline over 25 years/longer term. WDMr[1]: moderate non-breeding range decline over 25 years. BR/WR: breeding/non-breeding rarity. BL/WL: breeding/non-breeding localisation. Superscript text indicates whether species qualified as localised in IBAs, SPAs, or both. BI/WI: breeding/non-breeding international importance. Figures are given in bands for species exceeding the qualifying thresholds for the localisation and international importance criteria.

When a species has changed list since *BoCC3*, shading of table cells is used to indicate the criterion/criteria responsible for that change.

Eurasian Bittern moved from Red to Amber since it no longer qualifies under historical decline.

Table 4. Species Green-listed by *BoCC4*.

Name	*BoCC3*[a]	Name	*BoCC3*[a]
Tufted Duck *Aythya fuligula*	A[1]	Jackdaw *Corvus monedula*	G
Red-breasted Merganser *Mergus serrator*	G	Rook *Corvus frugilegus*	G
Goosander *Mergus merganser*	G	Carrion Crow *Corvus corone*	G
Ptarmigan *Lagopus muta*	G	Hooded Crow *Corvus cornix*	G
Red-throated Diver *Gavia stellata*	A[1]	Common Raven *Corvus corax*	G
Great Shearwater *Puffinus gravis*	G	Goldcrest *Regulus regulus*	G
Sooty Shearwater *Puffinus griseus*	A[1]	Firecrest *Regulus ignicapilla*	A[6]
Great Cormorant *Phalacrocorax carbo*	G	Blue Tit *Cyanistes caeruleus*	G
Little Egret *Egretta garzetta*	A[2]	Great Tit *Parus major*	G
Grey Heron *Ardea cinerea*	G	Crested Tit *Lophophanes cristatus*	A[1]
Little Grebe *Tachybaptus ruficollis*	A[3,4]	Coal Tit *Periparus ater*	G
Great Crested Grebe *Podiceps cristatus*	G	Bearded Tit *Panurus biarmicus*	A[7,2]
Red Kite *Milvus milvus*	A[1]	Woodlark *Lullula arborea*	A[1,8,2]
Northern Goshawk *Accipiter gentilis*	G	Sand Martin *Riparia riparia*	A[1]
Eurasian Sparrowhawk *Accipiter nisus*	G	Barn Swallow *Hirundo rustica*	A[1]
Common Buzzard *Buteo buteo*	G	Cetti's Warbler *Cettia cetti*	G
Golden Eagle *Aquila chrysaetos*	A[1]	Long-tailed Tit *Aegithalos caudatus*	G
Water Rail *Rallus aquaticus*	G	Common Chiffchaff *Phylloscopus collybita*	G
Moorhen *Gallinula chloropus*	G	Blackcap *Sylvia atricapilla*	G
Common Coot *Fulica atra*	G	Garden Warbler *Sylvia borin*	G
European Golden Plover *Pluvialis apricaria*	A[5]	Lesser Whitethroat *Sylvia curruca*	G
Little Ringed Plover *Charadrius dubius*	G	Common Whitethroat *Sylvia communis*	A[4]
Little Stint *Calidris minuta*	G	Sedge Warbler *Acrocephalus schoenobaenus*	G
Jack Snipe *Lymnocryptes minimus*	A[1]	Reed Warbler *Acrocephalus scirpaceus*	G
Pomarine Skua *Stercorarius pomarinus*	G	Waxwing *Bombycilla garrulus*	G
Long-tailed Skua *Stercorarius longicaudus*	G	Eurasian Nuthatch *Sitta europaea*	G
Little Auk *Alle alle*	G	Eurasian Treecreeper *Certhia familiaris*	G
Black Tern *Chlidonias niger*	A[1]	Wren *Troglodytes troglodytes*	G
Little Gull *Hydrocoloeus minutus*	A[1]	Blackbird *Turdus merula*	G
Rock Dove *Columba livia*	G	Robin *Erithacus rubecula*	G
Wood Pigeon *Columba palumbus*	G	European Stonechat *Saxicola rubicola*	G
Collared Dove *Streptopelia decaocto*	G	Northern Wheatear *Oenanthe oenanthe*	A[1]
Barn Owl *Tyto alba*	A[1]	Pied Wagtail *Motacilla alba*	G
Long-eared Owl *Asio otus*	G	Rock Pipit *Anthus petrosus*	G
Green Woodpecker *Picus viridis*	A[1]	Brambling *Fringilla montifringilla*	G
Great Spotted Woodpecker *Dendrocopos major*	G	Common Chaffinch *Fringilla coelebs*	G
Hobby *Falco subbuteo*	G	Greenfinch *Chloris chloris*	G
Peregrine Falcon *Falco peregrinus*	G	Common Crossbill *Loxia curvirostra*	G
Red-billed Chough *Pyrrhocorax pyrrhocorax*	A[1]	Goldfinch *Carduelis carduelis*	G
Magpie *Pica pica*	G	Siskin *Spinus spinus*	G
Eurasian Jay *Garrulus glandarius*	G		

[a] *BoCC3* assessments:
R = Red, A = Amber, G = Green. For species which have changed list since *BoCC3* (all of which have moved from the Amber list), the superscript text indicates which criteria they no longer qualify for Amber under. [1] = ERLOB (previously SPEC), [2] = breeding localisation, [3] = moderate breeding population decline over 25 years, [4] = moderate breeding population decline over longer term, [5] = non-breeding international importance, [6] = breeding rarity, [7] = moderate breeding range decline over 25 years, [8] = moderate breeding range decline over longer term.

Table 5. Number of species moving between Red, Amber and Green lists since *BoCC3*.

		BoCC4 status				
		Red	Amber	Green	Not assessed	Total
BoCC3 status	Red	47	3	0	2	52
	Amber	18	85	22	1	126
	Green	2	7	59	0	68
	Not assessed	0	1	0	–	1
	Total	67	96	81	3	247[1]

[1] Number of species assessed across *BoCC3* and *BoCC4* combined; *BoCC3* assessed 246 species, *BoCC4* 244 species.

lists since *BoCC3*, with 52 species (21% of those reassessed) changing *BoCC* status (table 5). The Red list has increased by 15, owing to 19 species being Red-listed for the first time, one species (Merlin *Falco columbarius*) returning to the Red list, and five species leaving the Red list either by moving to Amber (three species) or the list of former breeders (two). Of the species Red-listed for the first time, two moved directly from the Green list: White-fronted Goose *Anser albifrons* on account of the non-breeding population decline and Long-tailed Duck *Clangula hyemalis* as a consequence of being classified as Globally Threatened.

After a long decline from the nineteenth century onwards, the Wryneck last bred in the UK in 2002 and should now be considered a former breeder. Of the species to have been lost from the UK in modern times, this is probably the first that can be described as once having been common and widespread; it was recorded breeding in 54 counties between 1875 and 1900 (Holloway 1996). The other two species to have ceased breeding, Temminck's Stint and European Serin, have only ever been known as extremely rare or occasional breeders here.

The other notable change is the decrease in the relative length of the Amber list, which held 126 species in *BoCC3* but 96 in *BoCC4*:



Edmund Fellowes/BTO

419. The Merlin *Falco columbarius* returns to the Red list after being Amber-listed in *BoCC2* and *BoCC3*, as its recovery from historical decline has faltered.



Roger Riddington

420. Once a widespread breeding bird in the UK, the Wryneck *Jynx torquilla* is now classed as a former breeder, the last confirmed breeding record being in 2002. British birdwatchers can now expect to see it only as a spring and autumn passage migrant.

22 species moved from Amber to Green and 18 to Red, although seven were gained from the Green list and three from the Red. The net increase in the length of the Green list, by 14 species, is ostensibly good news and in some instances due to genuine improvements in the status of species, but see below for a discussion of the influence of changes in the assessment process, which has resulted in an estimated nine species moving to the Green list. The only new species assessed by *BoCC4*, Caspian Gull, went onto the Amber list. Table 5 summarises the movements between the three lists since *BoCC3*. Of the 243 species assessed by both *BoCC3* and *BoCC4*, 26 (10.7%) moved to a higher level of conservation concern and another 25 (10.3%) moved to a lower level of concern; the remaining 192 species (79.0%) did not change status between the two assessments.

An analysis of the reasons why species were Red-listed (which Red-list criteria they met) revealed that breeding population decline was by far the most important criterion; 50 species (74.6% of the Red list) qualified owing to declines over 25 years (12 species), the longer term (14) or both time periods (24). Fig. 1 shows how many species were listed against each criterion, and reveals that a significant number (21 species, over the two time periods) qualified for the Red list under no other criteria. Only six species qualified under non-breeding population decline, over either time period.

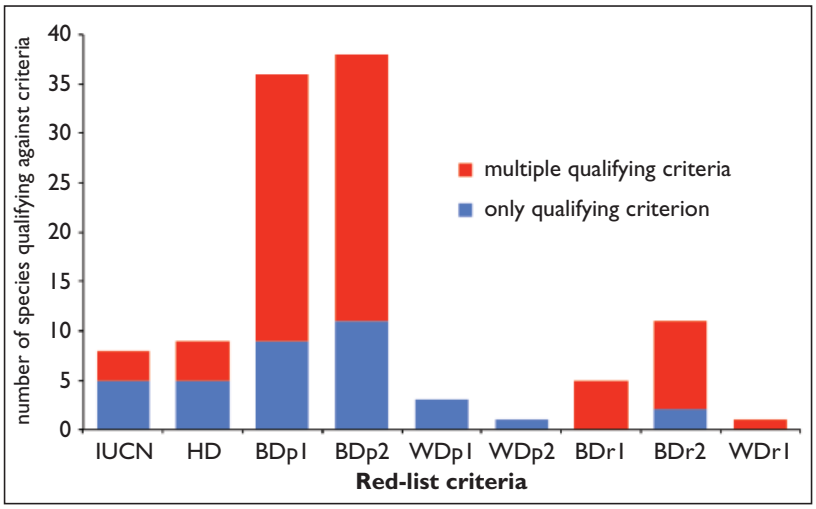


Fig. 1. Criteria under which species qualified for the *BoCC4* Red list. Bars show the number of species qualifying against each Red-list criterion: blue sections indicate the number of species which qualified against no other Red-list criteria, i.e. this criterion was the sole reason for the Red-listing.

The availability of new atlas data (Balmer *et al.* 2013) meant that the

range-change criterion increased in importance in this review. Whereas *BoCC3* listed only five species against severe range decline (all over the longer-term period), *BoCC4* found that 14 species showed a severe decline in range over at least one of the time periods and, notably, two species (Woodcock *Scolopax rusticola* and Cirl Bunting *Emberiza cirlus*) were Red-listed owing to range decline alone.

Finally, a concerning trend is the increasing number of the UK's species which are considered Globally Threatened. Whereas previous *BoCC* assessments have listed only two species, Balearic Shearwater and Aquatic Warbler, because of global threat, *BoCC4* lists an additional six: Common Pochard *Aythya ferina*, Long-tailed Duck, Velvet Scoter *Melanitta fusca*, Slavonian Grebe, Puffin *Fratercula arctica* and Turtle Dove *Streptopelia turtur*. Five of these eight Globally Threatened species did not qualify for Red-listing under any other criteria (Pochard, Slavonian Grebe and Turtle Dove being the exceptions).

The impact of changes in the assessment process

Although the *BoCC4* review has seen a substantial change in the composition of Red, Amber and Green lists, as described previously there were some changes in the way the review was conducted. We have explored the likely impact of these changes on our results, to be confident that the trends in list lengths are not an artefact of these changes.

We can clearly identify how our changes in treatment of recovery from historical decline (criteria HD and HDrec) influence the *BoCC4* outcome (table 6): if we had applied the approach used in *BoCC3,* then Merlin would have remained Amber-listed, under the HDrec criterion, rather than returning to Red as HD. Marsh Harrier and Osprey were considered to have shown complete recovery from historical decline by *BoCC3* but under *BoCC4* they are no longer considered to have recovered sufficiently to meet our new threshold. The change in assessment process is not responsible for a change in their *BoCC* status, however, as both species are also Amber-listed under additional criteria. The changes have no effect on the listing of other HD species.

The availability of wintering range data from *Bird Atlas 2007–11* meant that we were able to assess non-breeding range change (WDr[1]) across all relevant species for the first time. Very few species showed substantial non-breeding range declines; only one, Capercaillie *Tetrao urogallus,* declined by more than 50%, and no species were Red- or



Roger Riddington

421. Several results from the current review show the impact of a changing climate, and the movement of Ringed Plover *Charadrius hiaticula* from Amber to Red is one example, reflecting the decreasing number of winter visitors as birds are no longer pushed across to the UK by cold weather on the Continent



Ben Andrew/RSPB-images

422. Largely as a result of targeted conservation effort, to create and maintain reedbeds in suitable condition, the Eurasian Bittern *Botaurus stellaris* moved from Red to Amber in the current review, another step on its continued recovery as a breeding species in the UK.

Amber-listed on this criterion alone.

The change in how status at the European level was incorporated (moving from the SPEC to the ERLOB criterion) has had more of an impact on our lists, although it affects only potential listing on the Amber and Green lists. Some 65 species that qualified under the SPEC criterion in *BoCC3* did not qualify under ERLOB in the new assessment (only 20 UK species were listed as threatened by ERLOB), and as a consequence, 15 of these moved to the Green list (the remaining 50 being retained as Red or Amber through other criteria). Without having new SPEC assessments for comparison, it is difficult to be certain how many of those 15 species would have been retained on the Amber list if new SPECs had been available. Additional analyses conducted on data from EU member states (BirdLife International 2015) suggest that, at that scale, six species (e.g. Tufted Duck *Aythya fuligula* and Green Woodpecker *Picus viridis*) had recovered from the measures of population decline and/or depletion that resulted in them being SPEC-listed previously, and would not have been SPEC-listed if such assessments had been made. It is less clear for the remaining nine species, but it seems likely that most if not all of these would have been retained as SPEC – for example, Golden Eagle *Aquila chrysaetos* would have still qualified as Rare within Europe (see BirdLife International 2004). It is also possible that new assessment would have led to the SPEC-listing of some species for the first time, and potentially the movement of these species from the Green list to Amber.

In conclusion, the changes in *BoCC4* criteria resulted in one additional species on the Red list, and approximately nine additional species on the Green list, compared with the same criteria used for *BoCC3* (table 6).

Table 6. The likely impact of the changes of assessment criteria (for historical decline and European status) on *BoCC4* results.

Change in process	Effect (*BoCC3* list → *BoCC4* list)	Species affected
Changes in recovery from HD	Amber → Red	Merlin
Using ERLOB instead of SPEC status	Amber → Green	Sooty Shearwater, Golden Eagle, Jack Snipe, Black Tern, Little Gull, Red-billed Chough, Sand Martin, Barn Swallow, Northern Wheatear

Data gaps

We lacked population trends for 21 breeding species, including seabird species for which the UK is internationally important (e.g. Manx Shearwater *Puffinus puffinus*), upland species (e.g. Dunlin *Calidris alpina*), the endemic Scottish Crossbill *Loxia scotica* and a disparate collection of other species with distributions, habitat preferences and behaviours which mean that they elude the attentions of standard monitoring programmes (e.g. Eurasian Wigeon *Anas penelope*, Long-eared Owl *Asio otus* and Rock Pipit *Anthus petrosus*). These are important gaps, not least because, as shown in fig. 1, the criteria for breeding population decline tend to be by far the most influential in determining listing status. It is worth noting that another of this group, the Short-eared Owl *Asio flammeus*, showed a long-term decline in range of 47%; had population monitoring been undertaken for the same period it is distinctly possible that it may have qualified for the Red list. Noting that longer-term breeding trends were lacking for a much larger number of birds (54 species), however, does indicate that recent decades have seen a welcome improvement in our monitoring coverage.

Race-level assessment

BoCC4 assessments were made for 224 races (of 173 species) occurring regularly in the UK. Of these, 57 races (25.4%) were Red-listed, 94 (42.0%) Amber-listed, and 73 (32.6%) Green-listed; these proportions are similar to those for the species-level assessment. Lists of races on the three lists and the criteria under which they qualify are given in tables 7–9.

Eighteen races have moved onto the Red list since *BoCC3*: 16 from Amber, and two newly assessed races (Slavonian Grebe and Greater Scaup). Many of the moves to the Red list mirror changes in parent species, for example because of UK population declines which apply to the race as well as to the species, such as for Shag *Phalacrocorax a. aristotelis* and Pied Flycatcher *Ficedula h. hypoleuca*. However, three of the new Red-listed races are not Red-listed at species level (in all, 44 races have a different *BoCC4* listing from their parent species) including, most notably, the British race of Greenfinch *Chloris chloris harrisoni* – as a species, Greenfinch is Green-listed but the race would qualify as Globally Threatened due to recent decline, driven by outbreaks of the parasitic disease trichomonosis (Lawson *et al.* 2012).



Andy Hay/RSPB-images

423. The European Nightjar *Caprimulgus europaeus* moves from Red to Amber in *BoCC4*, joining other largely heathland and grassland species, such as Stone-curlew *Burhinus oedicnemus* and Woodlark *Lullula arborea*, which made the same move in the *BoCC3* review.



Ben Hall/RSPB-images

424. Common Pochard *Aythya ferina* has moved from Amber to Red as a consequence of population decline – not just in the UK, where it has shown a severe drop in non-breeding numbers, but also more widely. This international decline has resulted in it being listed as Vulnerable on the IUCN Global Red list.



Stanley Porter/RSPB-images

425. The Woodcock *Scolopax rusticola* moves from Amber to Red in *BoCC4* as a consequence of a shrinking breeding range in the UK. It is one of just two species (Cirl Bunting *Emberiza cirlus* being the other) that are Red-listed owing to range decline alone.

Table 7. Races on the *BoCC4* Red list and the criteria under which they qualify. Red[b] and Amber[c] criteria

Species[d] / Race	*BoCC4* species[a]	IUCN	HD	BD_{P1} & BDM_{P1}	BD_{P}^{2} & BDM_{P2}	WD_{P1} & WDM_{P1}	WD_{P}^{2} & WDM_{P2}	BD_{r1} & BDM_{r1}	BD_{r}^{2} & BDM_{r2}	WD_{r1} & WDM_{r1}	ERLOB	HD_{rec}	BR	WR	BL	WL	BI	WI
'Taiga Bean Goose' *Anser f. fabalis*	A	⋆									⋆			⋆				
'Greenland White-fronted Goose' *Anser albifrons flavirostris*	R	⋆									⋆							⋆
'European White-fronted Goose' *Anser a. albifrons*	R					⋆	⋆											
⋆Greater Scaup *Aythya m. marila*	R	⋆				⋆	⋆				⋆					⋆		
Common Eider *Somateria m. mollissima*	A	⋆									⋆							
Black Grouse *Tetrao tetrix britannicus*	R	⋆	⋆	⋆	⋆				⋆		⋆						⋆	
Capercaillie *Tetrao u. urogallus*	R		⋆	⋆	⋆				⋆	⋆								
Grey Partridge *Perdix p. perdix*	R			⋆	⋆				⋆						⋆			
⋆Fulmar *Fulmarus g. glacialis*	A	⋆									⋆							
⋆Shag *Phalacrocorax a. aristotelis*	R			⋆	⋆									⋆			⋆	
⋆Red-necked Grebe *Podiceps g. grisegena*	R					⋆								⋆				
⋆Slavonian Grebe *Podiceps a. auritus*	R			⋆	⋆			⋆					⋆					
Hen Harrier *Circus c. cyaneus*	R		⋆															
⋆Ringed Plover *Charadrius h. hiaticula*	R			⋆		⋆	⋆									⋆		
Whimbrel *Numenius p. phaeopus*	R			⋆				⋆	⋆									
⋆Eurasian Curlew *Numenius a. arquata*	R			⋆	⋆						⋆						⋆	
Black-tailed Godwit *Limosa l. limosa*	R	⋆	⋆					⋆			⋆							
Roseate Tern *Sterna d. dougallii*	R			⋆	⋆			⋆	⋆				⋆					
⋆Kittiwake *Rissa t. tridactyla*	R			⋆	⋆						⋆							
Herring Gull *Larus argentatus argenteus*	R	⋆		⋆	⋆						⋆						⋆	
Turtle Dove *Streptopelia t. turtur*	R	⋆		⋆	⋆				⋆		⋆							
Common Cuckoo *Cuculus c. canorus*	R			⋆	⋆													
⋆Common Kingfisher *Alcedo atthis ispida*	A	⋆			⋆						⋆							
Lesser Spotted Woodpecker *Dendrocopos minor comminutus*	R	⋆		⋆	⋆			⋆	⋆		⋆						⋆	
⋆Merlin *Falco columbarius aesalon*	R		⋆															

Table 7. Races on the *BoCC4* Red list and the criteria under which they qualify. Red[b] and Amber[c] criteria (cont.)

Species[d] / Race	*BoCC4* species[a]	IUCN	HD	BDp[1] & BDMp[1]	BDp[2] & BDMp[2]	WDp[1] & WDMp[1]	WDp[2] & WDMp[2]	BDr[1] & BDMr[1]	BDr[2] & BDMr[2]	WDr[1] & WDMr[1]	ERLOB	HDrec	BR	WR	BL	WL	BI	WI
Golden Oriole *Oriolus o. oriolus*	R			*	*			*					*					
Red-backed Shrike *Lanius c. collurio*	R		*	*	*				*				*					
Willow Tit *Poecile montana kleinschmidti*	R	*		*	*			*	*	*	*						*	
Marsh Tit *Poecile palustris dresseri*	R			*	*												*	
Skylark *Alauda a. arvensis*	R			*	*													
Grasshopper Warbler *Locustella n. naevia*	R			*	*													
Savi's Warbler *Locustella l. luscinioides*	R			*	*			*					*					
'Fair Isle Wren' *Troglodytes troglodytes fridariensis*	G	*									*		*				*	
'St Kilda Wren' *Troglodytes t. hirtensis*	G	*									*		*				*	
Common Starling *Sturnus v. vulgaris*	R			*	*													
Ring Ouzel *Turdus t. torquatus*	R			*					*									
'Hebridean Song Thrush' *Turdus philomelos hebridensis*	R	*									*						*	
Song Thrush *Turdus p. clarkei*	R				*												*	
Redwing *Turdis i. iliacus*	R			*	*			*	*				*					
*Mistle Thrush *Turdus v. viscivorus*	R			*	*													
Spotted Flycatcher *Muscicapa s. striata*	R			*	*													
Common Nightingale *Luscinia m. megarhynchos*	R			*	*				*									
*Pied Flycatcher *Ficedula h. hypoleuca*	R			*				*										
*Black Redstart *Phoenicurus ochruros gibraltariensis*	R			*	*								*	*				
House Sparrow *Passer d. domesticus*	R			*	*													
Tree Sparrow *Passer m. montanus*	R				*				*									
Yellow Wagtail *Motacilla flava flavissima*	R	*		*	*			*	*		*						*	
*Grey Wagtail *Motacilla c. cinerea*	R			*	*													
Tree Pipit *Anthus t. trivialis*	R			*	*				*									

Table 7. Races on the *BoCC4* Red list and the criteria under which they qualify. Red[b] and Amber[c] criteria (cont.)

Species[d] / Race	*BoCC4* species[a]	IUCN	HD	BDp¹ & BDMp¹	BDp² & BDMp²	WDp¹ & WDMp¹	WDp² & WDMp²	BDr¹ & BDMr¹	BDr² & BDMr²	WDr¹ & WDMr¹	ERLOB	HDrec	BR	WR	BL	WL	BI	WI
Hawfinch *Coccothraustes c. coccothraustes*	R			*				*	*									
*Greenfinch *Chloris chloris harrisoni*	G	*									*						*	
Linnet *Linaria c. cannabina*	R				*													
Twite *Linaria flavirostris bensonorum*	R	*		*													*	
Twite *Linaria f. pipilans*	R		*	*													*	
Yellowhammer *Emberiza c. citrinella*	R			*	*													
Corn Bunting *Emberiza calandra clanceyi*	R	*	*	*	*						*		*				*	
Corn Bunting *Emberiza c. calandra*	R		*	*	*			*	*	*								

[a] *BoCC4* assessments for 'parent' species: R = Red, A = Amber, G = Green

[b] Red-list criteria:

IUCN: Globally Threatened. HD: historical decline in the breeding population. BDp¹/²: severe breeding population decline over 25 years/longer term. WDp¹/²: severe non-breeding population decline over 25 years/longer term. BDr¹/²: severe breeding range decline over 25 years/longer term. WDr¹: severe non-breeding range decline over 25 years.

[c] Amber-list criteria:

ERLOB: Threatened in Europe. HDrec: historical decline – recovery. BDMp¹/²: moderate breeding population decline over 25 years/longer term. WDMp¹/²: moderate non-breeding population decline over 25 years/longer term. BDMr¹/²: moderate breeding range decline over 25 years/longer term. WDMr¹: moderate non-breeding range decline over 25 years. BR/WR: breeding/non-breeding rarity. BL/WL: breeding/non-breeding localisation. BI/WI: breeding/non-breeding international importance.

Red and Amber criteria for population and range trends are combined in the same columns (e.g. BDp¹ and BDMp¹): red and amber colour-coding is used to show which a species qualified against.

[d] Asterisks indicate species with a changed race-level status since *BoCC3*.

This table lists Red-listed races of polytypic species only: it does not include monotypic species, e.g. Long-tailed Duck.

Table 8. Races on the *BoCC4* Amber list and the criteria under which they qualify.

Species[c] / Race	*BoCC4* species[a]	ERLOB	HDrec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	WDMr[1]	BDMr[2]	BR	WR	BL	WL	BI	WI
Bewick's Swan *Cygnus columbianus bewickii*	A	*				*								*		*
'Tundra Bean Goose' *Anser fabalis rossicus*	A											*				
Greylag Goose *Anser a. anser*	A													*		*
*'Dark-bellied Brent Goose' *Branta b. bernicla*	A													*		*
'Pale-bellied Brent Goose' *Branta b. hrota*	A													*		*
Eurasian Teal *Anas c. crecca*	A															*
Mallard *Anas p. platyrhynchos*	A					*										
Common Eider *Somateria mollissima faeroeensis*	A			*	*										*	
Common Goldeneye *Bucephala c. clangula*	A										*					
Common Quail *Coturnix c. coturnix*	A		*													
Red Grouse *Lagopus lagopus scotica*	A														*	
Ptarmigan *Lagopus muta millaisi*	G														*	
Black-throated Diver *Gavia a. arctica*	A										*	*				
European Storm-petrel *Hydrobates p. pelagicus*	A												*			
Leach's Storm-petrel *Oceanodroma l. leucorhoa*	A												*		*	
Great Cormorant *Phalacrocorax c. carbo*	G														*	
Great Cormorant *Phalacrocorax carbo sinensis*	G												*			
Eurasian Bittern *Botaurus s. stellaris*	A		*								*	*				
Eurasian Spoonbill *Platalea l. leucorodia*	A										*	*				
Black-necked Grebe *Podiceps n. nigricollis*	A										*	*				
Marsh Harrier *Circus a. aeruginosus*	A		*										*			
Osprey *Pandion h. haliaetus*	A		*								*					
Common Crane *Grus g. grus*	A										*	*				
*Stone-curlew *Burhinus o. oedicnemus*	A									*			*			
Oystercatcher *Haematopus o. ostralegus*	A	*												*	*	*
Black-tailed Godwit *Limosa limosa islandica*	R										*			*		
Bar-tailed Godwit *Limosa l. lapponica*	A	*												*		*
Turnstone *Arenaria i. interpres*	A					*										*

Table 8. Races on the *BoCC4* Amber list and the criteria under which they qualify. (cont.)

Species[c] / Race	*BoCC4* species[a]	ERLOB	HDrec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	WDMr[1]	BDMr[2]	BR	WR	BL	WL	BI	WI
Red Knot *Calidris canutus islandica*	A													★		★
Dunlin *Calidris a. alpina*	A						★							★		★
*Dunlin *Calidris alpina schinzii*	A							★					★			
Common Redshank *Tringa t. totanus*	A			★		★		★		★						
Common Redshank *Tringa t. robusta*	A					★										★
Common Snipe *Gallinago g. gallinago*	A									★						
Common Snipe *Gallinago g. faeroeensis*	A															★
*Black Guillemot *Cepphus grylle arcticus*	A							★								
Razorbill *Alca torda islandica*	A												★		★	
Common Guillemot *Uria a. aalge*	A												★			
Common Guillemot *Uria a. albionis*	A												★		★	
Little Tern *Sternula a. albifrons*	A									★			★			
Common Tern *Sterna h. hirundo*	A												★			
Common Gull *Larus c. canus*	A															★
Lesser Black-backed Gull *Larus fuscus graellsii*	A														★	
Glaucous Gull *Larus h. hyperboreus*	A											★				
Iceland Gull *Larus g. glaucoides*	A											★				
Yellow-legged Gull *Larus m. michahellis*	A										★					
Stock Dove *Columba o. oenas*	A														★	
*Tawny Owl *Strix aluco sylvatica*	A			★	★											
Short-eared Owl *Asio f. flammeus*	A							★		★						
European Nightjar *Caprimulgus e. europaeus*	A									★						
Common Swift *Apus a. apus*	A			★												
Great Spotted Woodpecker *Dendrocopos major anglicus*	G														★	
Common Kestrel *Falco t. tinnunculus*	A			★	★											
Merlin *Falco columbarius subaesalon*	R															★
Red-billed Chough *Pyrrhocorax p. pyrrhocorax*	G														★	
Eurasian Jay *Garrulus glandarius hibernicus*	G														★	

Table 8. Races on the *BoCC4* Amber list and the criteria under which they qualify. (cont.)

Species[c] / Race	*BoCC4* species[a]	ERLOB	HDrec	BDMp[1]	BDMp[2]	WDMp[1]	WDMp[2]	BDMr[1]	WDMr[1]	BDMr[2]	BR	WR	BL	WL	BI	WI
Eurasian Jay *Garrulus g. rufitergum*	G														★	
Blue Tit *Cyanistes caeruleus obscurus*	G														★	
Great Tit *Parus major newtoni*	G														★	
Crested Tit *Lophophanes cristatus scoticus*	G														★	
Coal Tit *Periparus ater britannicus*	G			★											★	
Skylark *Alauda arvensis scotica*	R														★	
Shore Lark *Eremophila alpestris flava*	A											★				
House Martin *Delichon u. urbicum*	A			★	★											
Long-tailed Tit *Aegithalos caudatus rosaceus*	G														★	
Willow Warbler *Phylloscopus t. trochilus*	A			★	★											
Dartford Warbler *Sylvia undata dartfordiensis*	A		★										★			
Short-toed Treecreeper *Certhia brachydactyla megarhyncha*	A										★					
'Hebridean Wren' *Troglodytes troglodytes hebridensis*	G														★	
'Shetland Wren' *Troglodytes t. zetlandicus*	G														★	
Wren *Troglodytes troglodytes indigenus*	G														★	
'Shetland Starling' *Sturnus vulgaris zetlandicus*	R														★	
Dipper *Cinclus cinclus gularis*	A			★											★	
Dipper *Cinclus c. hibernicus*	A														★	
Common Redstart *Phoenicurus p. phoenicurus*	A									★						
Dunnock *Prunella modularis occidentalis*	A				★										★	
'Blue-headed Wagtail' *Motacilla f. flava*	R										★	★				
'Grey-headed Wagtail' *Motacilla f. thunbergi*	R											★				
Pied Wagtail *Motacilla alba yarrellii*	G														★	
*'White Wagtail' *Motacilla a. alba*	G										★					
Meadow Pipit *Anthus p. pratensis*	A				★											
Meadow Pipit *Anthus p. whistleri*	A														★	
Water Pipit *Anthus s. spinoletta*	A											★				
Rock Pipit *Anthus p. petrosus*	G														★	

Table 8. Races on the *BoCC4* Amber list and the criteria under which they qualify. (cont.)

Species[c] / Race	*BoCC4* species[a]	ERLOB	HDrec	$BDMp^1$	$BDMp^2$	$WDMp^1$	$WDMp^2$	$BDMr^1$	$WDMr^1$	$BDMr^2$	BR	WR	BL	WL	BI	WI
Common Chaffinch *Fringilla coelebs gengleri*	G														⋆	
Bullfinch *Pyrrhula pyrrhula pileata*	A				⋆										⋆	
Linnet *Linaria cannabina autochthona*	R			⋆											⋆	
⋆Common Redpoll *Acanthis f. flammea*	A											⋆				
Goldfinch *Carduelis carduelis britannica*	G														⋆	
Snow Bunting *Plectrophenax n. nivalis*	A										⋆					
Snow Bunting *Plectrophenax nivalis insulae*	A										⋆					
Lapland Bunting *Calcarius l. lapponicus*	A											⋆				
Lapland Bunting *Calcarius l. subcalcaratus*	A											⋆				
Reed Bunting *Emberiza s. schoeniclus*	A				⋆											

[a] *BoCC4* assessments for 'parent' species: R = Red, A = Amber, G = Green

[b] Amber-list criteria:

ERLOB: Threatened in Europe. HDrec: historical decline – recovery. $BDMp^{1/2}$: moderate breeding population decline over 25 years/longer term. $WDMp^{1/2}$: moderate non-breeding population decline over 25 years/longer term. $BDMr^{1/2}$: moderate breeding range decline over 25 years/longer term. $WDMr^1$: moderate non-breeding range decline over 25 years. BR/WR: breeding/non-breeding rarity. BL/WL: breeding/non-breeding localisation. BI/WI: breeding/non-breeding international importance.

[c] Asterisks indicate species with a changed race-level status since *BoCC3*.

This table lists Amber-listed races of polytypic species only: it does not include monotypic species, e.g. Pink-footed Goose *Anser brachyrhynchus*.

Table 9. Races on the *BoCC4* Green list.

Species / Race	*BoCC4* species[a]	Species / Race	*BoCC4* species[a]
Goosander *Mergus m. merganser*	G	Goldcrest *Regulus r. regulus*	G
Little Egret *Egretta g. garzetta*	G	Firecrest *Regulus i. ignicapilla*	G
Grey Heron *Ardea c. cinerea*	G	Blue Tit *Cyanistes c. caeruleus*	G
Little Grebe *Tachybaptus r. ruficollis*	G	Great Tit *Parus m. major*	G
Great Crested Grebe *Podiceps c. cristatus*	G	Coal Tit *Periparus a. ater*	G
Northern Goshawk *Accipiter g. gentilis*	G	Coal Tit *Periparus a. hibernicus*	G
Eurasian Sparrowhawk *Accipiter n. nisus*	G	Bearded Tit *Panurus b. biarmicus*	G
Common Buzzard *Buteo b. buteo*	G	Woodlark *Lullula a. arborea*	G
Golden Eagle *Aquila c. chrysaetos*	G	Sand Martin *Riparia r. riparia*	G
Water Rail *Rallus a. aquaticus*	G	Barn Swallow *Hirundo r. rustica*	G
Moorhen *Gallinula c. chloropus*	G	Cetti's Warbler *Cettia c. cetti*	G
Common Coot *Fulica a. atra*	G	Common Chiffchaff *Phylloscopus c. collybita*	G
Little Ringed Plover *Charadrius dubius curonicus*	G	Willow Warbler *Phylloscopus trochilus acredula*	A
Ringed Plover *Charadrius hiaticula tundrae*	R	Blackcap *Sylvia a. atricapilla*	G
Dunlin *Calidris alpina arctica*	A	Garden Warbler *Sylvia b. borin*	G
Long-tailed Skua *Stercorarius l. longicaudus*	G	Lesser Whitethroat *Sylvia c. curruca*	G
Razorbill *Alca t. torda*	A	Common Whitethroat *Sylvia c. communis*	G
Little Auk *Alle a. alle*	G	Reed Warbler *Acrocephalus s. scirpaceus*	G
Black Tern *Chlidonias n. niger*	G	Waxwing *Bombycilla g. garrulus*	G
Lesser Black-backed Gull *Larus fuscus intermedius*	A	Eurasian Nuthatch *Sitta europaea caesia*	G
Herring Gull *L. a. argentatus*	R	Eurasian Treecreeper *Certhia familiaris britannica*	G
Rock Dove/Feral Pigeon *Columba l. livia*	G	Wren *Troglodytes t. troglodytes*	G
Wood Pigeon *Columba p. palumbus*	G	Blackbird *Turdus m. merula*	G
Collared Dove *Streptopelia d. decaocto*	G	Song Thrush *Turdus p. philomelos*	R
Barn Owl *Tyto a. alba*	G	Redwing *Turdus iliacus coburni*	R
Long-eared Owl *Asio o. otus*	G	Robin *Erithacus r. rubecula*	G
Green Woodpecker *Picus v. viridis*	G	Robin *Erithacus r. melophilus*	G
Hobby *Falco s. subbuteo*	G	European Stonechat *Saxicola rubicola hibernans*	G
Peregrine Falcon *Falco p. peregrinus*	G	Northern Wheatear *Oenanthe o. oenanthe*	G
Magpie *Pica p. pica*	G	'Greenland Wheatear' *Oenanthe o. leucorhoa*	G
Eurasian Jay *Garrulus g. glandarius*	G	Dunnock *Prunella m. modularis*	A
Jackdaw *Corvus m. monedula*	G	Dunnock *Prunella m. hebridium*	A
Jackdaw *Corvus m. spermologus*	G	Rock Pipit *Anthus petrosus littoralis*	G
Rook *Corvus f. frugilegus*	G	Common Chaffinch *Fringilla c. coelebs*	G
Carrion Crow *Corvus c. corone*	G	Greenfinch *Chloris c. chloris*	G
Hooded Crow *Corvus c. cornix*	G	Common Crossbill *Loxia c. curvirostra*	G
Common Raven *Corvus c. corax*	G		

[a] *BoCC4* assessments for 'parent' species: R = Red, A = Amber, G = Green.
This table lists Green-listed races of polytypic species only: it does not include monotypic species, e.g. Brambling *Fringilla montifringilla*.

Discussion

The growing Red list

BoCC4 has placed more species onto the Red list than ever before. Some 67 species are Red-listed (27.5% of the species assessed) and that list has grown by a substantially larger increment than in any previous *BoCC* review (fig. 2). In total, 20 species have moved to Red, with only three species moving from Red to Amber.

The Red list increased substantially between the second and third *BoCC* reviews but a number of those additions were due to changes to the assessment process. In particular, the introduction of the longer-term time window for consideration of population and range trends resulted in 11 species moving to (or staying on) the Red list that would not otherwise have done so. This is not the case here; only Merlin has returned to the Red list as a consequence of changes to the way we treat recovery from historical decline. The other significant change in our process is the treatment of conservation concern at a European level because we lack a current SPEC assessment. This has resulted

in a number of species that may otherwise have been Amber-listed being moved to the Green list; the Green list grew by 13 species, of which nine (or possibly more) may have been Amber-listed had we been able to retain the use of SPEC. Some of these species continue to merit conservation attention, including Red-billed Chough *Pyrrhocorax pyrrhocorax*, which remains relatively rare and range-restricted in the UK; and Golden Eagle, also relatively rare and range-restricted in the UK, due to persecution, both historical and recent (Whitfield *et al.* 2007).

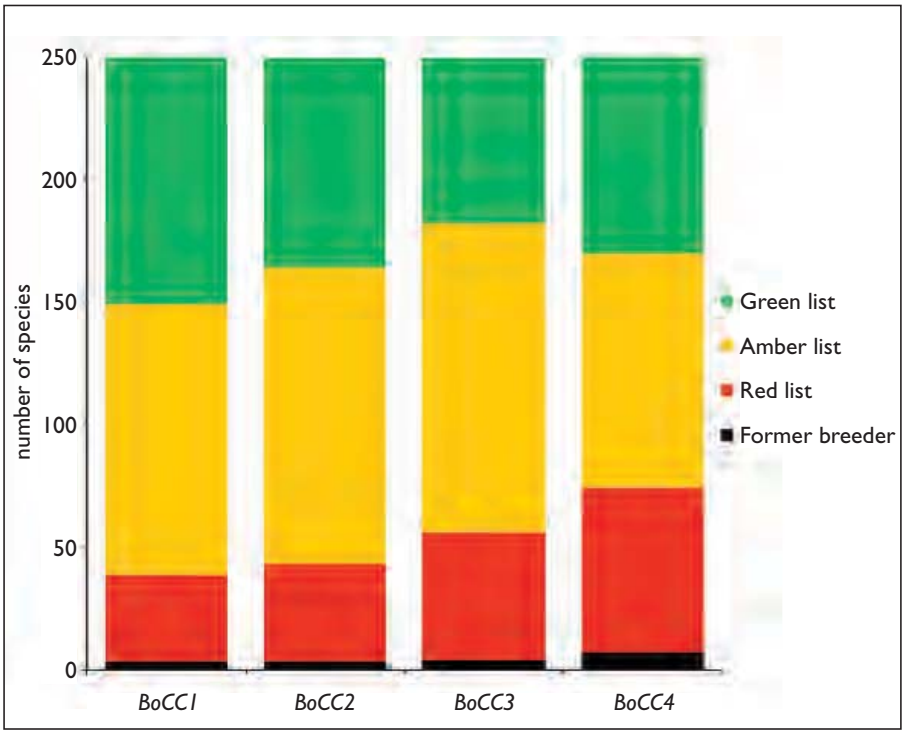


Fig. 2. Lengths of Red, Amber and Green lists in the four *BoCC* assessments. Note that the assessment process has developed over time, with changes in data availability and criteria between assessments, and a small number of changes in Red, Amber and Green list lengths have been as a consequence of these changes.

A priority list such as *BoCC4*, or a national IUCN Red List, should not, however, be the only consideration in decisions on which species should be the recipients of conservation effort. As well as *BoCC* status, we encourage the consideration of other factors, such as likelihood of conservation action being successful, the logistics of such action and synergies with other conservation activi-



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426. The Whinchat *Saxicola rubetra* moves from Amber to the Red list in *BoCC4*, and is a member of two distinct groupings to cause concern – upland species and Afro-Palearctic migrants.

ties. And while we might expect most Red-listed species to be the highest priorities for conservation, there are some on which it might not be appropriate to expend scarce conservation resources. These might include species at the edge of their European range in the UK, for which the factors that determine their abundance in the UK may lie elsewhere. Conversely, there are species on the *BoCC* Amber list that have been, and may continue to be, high priorities for conservation action, especially ones that might be considered as conservation dependent. There have been a number of noteworthy conservation successes in the UK due to the delivery of targeted and well-informed conservation action for priority bird species. While many of these, such as Corn Crake *Crex crex* and Cirl Bunting, remain Red-listed, we should celebrate the movement of others from Red to Amber, such as Red Kite and Marsh Harrier in *BoCC2*, Stone-curlew and Woodlark in *BoCC3*, and Eurasian Bittern *Botaurus stellaris* and European Nightjar *Caprimulgus europaeus* (as well as Red Kite moving to the Green list) in *BoCC4*. Simply because a species moves from Red to Amber does not, however, necessarily mean that conservation effort can be withdrawn immediately, as many remain dependent upon conservation action. A good example is the Stone-curlew. A large part of the UK population nests in arable fields, in which labour-intensive interventions are required to protect the birds from agricultural operations; an abrupt cessation of that effort would most likely result in the Stone-curlew's return to the Red list. Work is ongoing to encourage more birds to nest in semi-natural grasslands or in safe nesting plots on arable land, supported by agri-environment schemes, paving the way for a more sustainable population.

Themes in bird conservation in the UK, as highlighted by *BoCC4*

Some consistent themes have emerged from previous assessments and other overviews of the status of the UK's biodiversity (e.g. Burns *et al.* 2013), and this review largely reiterates these. Our overriding concern is for the ever-increasing number of species on the Red list: despite a proven ability to improve the status of species of concern, the rate at which species are added to the Red list greatly exceeds our current ability to take recovery action. If we believe that the presence of species on Red lists is an effective barometer of the state of our wildlife (e.g. Butchart *et al.* 2005), then this review paints a bleak picture.

In addition to the increase in the number of species on the Red list, three species have moved to the list of former breeders. Although this is loss at a UK rather than global scale, and while for highly mobile taxa such as birds recolonisation can never be ruled out, these losses should not be overlooked. In particular, Wryneck becomes the first once-widespread species to have been lost from the UK since the extinction of the Great Bustard *Otis*



Roger Riddington

427. Concerns about the state of the UK's internationally important seabird populations is heightened by the *BoCC4* review, with three familiar species moving to the Red list, the Puffin *Fratercula arctica* as a result of its IUCN listing as Vulnerable.

tarda in around 1833. It is a sobering thought that the Wryneck was once sufficiently common for the RSPB to sell nestboxes for it.

That no new farmland birds have moved to the Red list probably reflects the fact that the species which continue to be affected adversely by modern agricultural methods are already listed there. Although the trends of some of these species have levelled out in recent years, others continue to decline; most alarmingly in the case of Turtle Dove, which has declined by 13% per annum since 1995 (Harris *et al.* 2015). Declines in woodland specialists (as opposed to generalists, which on the whole have been doing well; Defra 2014) were highlighted in *BoCC3*, and this review adds three more woodland birds, Woodcock, Common Nightingale *Luscinia megarhynchos* and Pied Flycatcher, to the Red list. There are now 16 woodland species on the Red list, more than any other habitat group, although a higher proportion of farmland species are Red-listed (fig. 3).

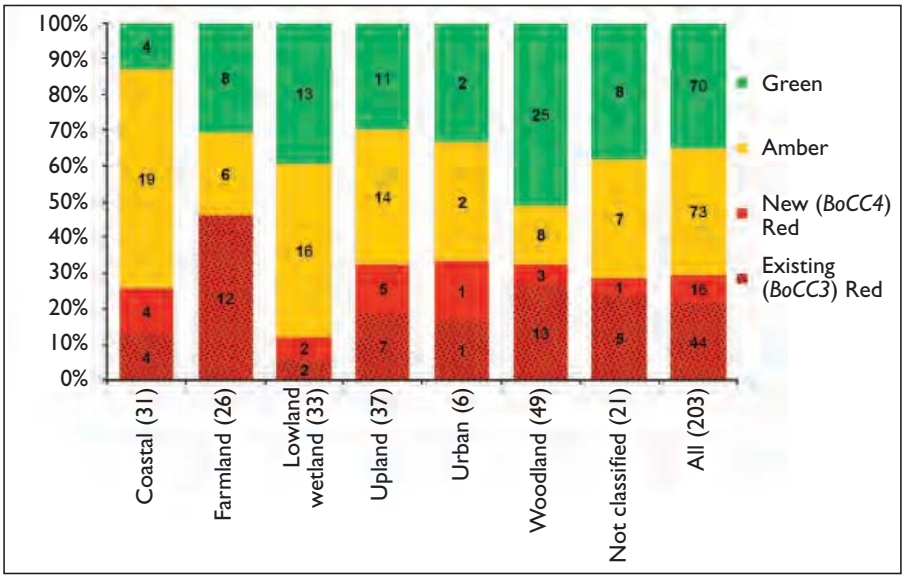


Fig. 3. Proportion of breeding birds in the Red, Amber and Green lists by major habitat type (habitat categories follow Gibbons *et al.* 1993). Bars show percentages in the Red, Amber and Green lists, figures give the actual number of species.

The greatest increases in the proportion of species Red-listed are for birds breeding in upland and coastal habitats (five and four



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428. Another woodland specialist and long-distance migrant, the Common Nightingale *Luscinia megarhynchos* shows such a severe decline in breeding numbers that it is now Red-listed.

species respectively). The increase in coastal species chiefly reflects the deteriorating status of the UK's seabirds; with the addition of Kittiwake *Rissa tridactyla*, Shag and Puffin, the number of seabirds on the Red list has nearly doubled. Furthermore, with Razorbill *Alca torda* now considered as globally Near Threatened (BirdLife International 2015), there is growing concern for our seabirds, particularly as in global terms they are among the most important components of the UK's avifauna. We should also note that, with the addition of Velvet Scoter and Long-tailed Duck, four of the UK's seaducks are now on the Red list, although the causes of their declines may be different from and possibly unrelated to marine impacts.

The recent *Bird Atlas 2007–11* (Balmer *et al.* 2013) highlighted two areas of concern that, arguably, have not before been recognised as being among the UK's highest conservation priorities: declines in the ranges of both breeding waders and upland breeding species (and there is, of course, much overlap between these two groups). *BoCC4* lends support for this view. The addition of five upland breeding species to the Red list – Dotterel, Eurasian Curlew *Numenius arquata*, Merlin, Whinchat *Saxicola rubetra* and Grey Wagtail *Motacilla cinerea* – means that there are as many species of upland birds Red-listed as there are farmland birds. In total, there are now nine species of wader on the Red list, and while the drivers of the declines are likely to be varied, it is clear that this group is under pressure (of 22 wader species breeding in the UK, only two remain on the Green list). Brown *et al.* (2015) argued that Eurasian Curlew should currently be considered the UK's most pressing bird conservation priority, given the global concern (Near Threatened) for the species, the significance of the UK's breeding population and the rapid decline in that population.

Another concern raised by the *BoCC3* assessment was population decline in a growing number of long-distance migrants, particularly those that winter in sub-Saharan Africa, and more specifically in the humid tropics (which have shown greater recent declines than species wintering in other regions; Hayhow *et al.* 2014). A further three Afro-Palearctic migrants, Common Nightingale, Pied Flycatcher and Whinchat, moved to the Red list in this review, and declines have continued in the majority of those listed already.



Andy Hay/RSPB-images

429. The UK holds about half the world's population of Greenland White-fronted Geese *Anser albifrons flavirostris* in winter. The most recent census results (2014/15) indicate the lowest numbers in Britain for 30 years. The ultimate causes of a collapse in productivity remains poorly understood, but probably relates to changing spring weather conditions and competition with Canada Geese *Branta canadensis* on the breeding areas – probably acting in combination.

Climate change may be behind some of the changes in listings reported here. Many species are thought to benefit from climate change (e.g. Pearce-Higgins *et al.* 2013), and the population increases in Little Egret *Egretta garzetta* and Firecrest *Regulus ignicapilla*, which have resulted in their move to the Green list, are likely to be at least partly in response to the UK's warming climate. Other species may be adversely affected by the UK's changing climate, including those at the southern edge of their range for which the 'climatic envelope' (the area within which climatic conditions are suitable for a species) is moving away from the UK (Huntley *et al.* 2007). This could be the case, for example, for Dotterel, although other pressures, such as increased nitrogen deposition and grazing, may have caused its decline (Hayhow *et al.* 2015). Other climate change impacts include the shifting of wintering ranges, which has led to UK population declines in White-fronted Goose and Ringed Plover *Charadrius hiaticula*, and the influence of climate upon marine food chains, which is affecting the food supplies of the Kittiwake (Frederiksen *et al.* 2007) and other seabirds.



Ben Hall/RSPB-images

430. The Greenfinch *Chloris chloris* is Green-listed as a species in this review, yet the British race *C. c. harrisoni* is Red-listed as a result of recent decline, driven by outbreaks of the parasitic disease trichomonosis.

BoCC at the race level

This was the second *BoCC* assessment to look at the status of regularly occurring races of birds in the UK, and we believe that they serve as a useful complement to the species-level assessments. We recommend that they are used to draw distinctions between the differing status of races of the same species, enabling better targeted conservation action – for example towards the nominate race of Black-tailed Godwit rather than the prospering Icelandic race *L. l. islandica*. In addition, we should highlight the precarious status of some races that are endemic, or nearly so, to the UK. While the loss of Wryneck as a UK breeding species is to be lamented, our birds were of the nominate race, which is still found widely across Europe; the rapidly declining British populations of Lesser Spotted Woodpecker *Dendrocopos minor comminutus* and Willow Tit *Poecile montana kleinschmidti* are of endemic races, so if lost would be gone forever.

The future of *BoCC*

While *BoCC* assessments provide a clear foundation for identifying priority bird species, this is not the only way of doing so, and indeed a different approach has been used to identify priority species for the UK's devolved administrations. Assessment against the *BoCC* criteria is rather a 'data-hungry' process, designed around the evidence available for birds, but it is simply not possible to replicate this approach for most other taxa, for which our knowledge is much poorer. This leaves birds as an exception to the growing practice of conducting national (usually for Great Britain, although sometimes for Britain and Ireland and occasionally for individual nations) Red List assessments using IUCN criteria (IUCN

2012). Burns *et al.* (2013) found British Red List assessments for 6,225 species of wildlife, but in the two years since then new assessments have been published, or are near publication, for many groups. It may be that while maintaining the series of *BoCC* assessments we also need to consider a national IUCN Red List assessment for birds, to enable a level playing field when assessing conservation priorities across all of the UK's biodiversity. We do, however, retain reservations about the regional IUCN Red Listing process, and the suitability of assessments focused on extinction risk alone for conservation prioritisation and action in the UK (see Eaton *et al.* 2005).

At present, *BoCC* and other priority-listing approaches are based solely on the current status of species, and give no consideration of likely future changes. We know that our environment is undergoing rapid changes, which will affect our bird populations for better or worse. For example, Huntley *et al.* (2007) used climate envelope modelling to show how the ranges of European breeding species were likely to move north and east in response to climate change by the late twenty-first century. As a consequence, we suspect that conditions in the UK might become more favourable for some species, but less favourable for others. Ausden *et al.* (2015) predicted which species are likely to be gained and lost as breeding species in the UK, forecasting the arrival of Short-toed Eagle *Circaetus gallicus* and Melodious Warbler *Hippolais polyglotta* among others, but also the climate-driven loss of breeding species such as Common Scoter *Melanitta nigra* and Pintail. This prompts the question of whether our priority setting should consider predicted future change, although it is not immediately clear how those predicted changes should be treated. Should we list species that have yet to begin breeding in the UK, to help ensure that we are ready for them when they do? After all, conserving those species for which lower latitudes are becoming less suitable is likely to become increasingly important.

The *BoCC* Red list is now lengthy, and contains a spread of species for which we have varying conservation concern. Some are considered to be under the threat of extinction globally, or are undergoing dramatic declines here that may lead to extinction in the UK – Willow Tit, Turtle Dove and Capercaillie, to name just three of the 19 species suggested as being at high risk of UK extinction by Ausden *et al.* (2015). Other Red-listed species, while still much-depleted from



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431. The Green Woodpecker *Picus viridis* is one of 22 species moving from Amber to Green, reflecting its improved status in Europe.

previous levels, have shown stable or even increasing trends in recent years, for example Song Thrush *Turdus philomelos*.

This fourth *BoCC* assessment now sits within the six-year cycle of reporting to the European Commission, and we anticipate future *BoCC* reviews remaining so. A timetable for EU reporting requirements, the production of new UK population estimates by APEP, and new European Red List assessments should enable us to produce the fifth *BoCC* in 2021. In the intervening period, it is vital that we maintain the monitoring programmes that *BoCC* relies upon, and continue to work with and support the UK's many thousands of dedicated birdwatchers to improve our evidence base. As mentioned previously, gaps in data remain, and while we are enduring lean times for the funding of conservation activities, we should strive to find efficient and imaginative ways of improving our monitoring to ensure that species do not slip through the net. Most importantly, we argue that there should be no let-up in our conservation action for the species most in need of it.

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Environment Agency

planning a sustainable approach



Minsmere Flood Risk Management Study

January 2009

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Introduction

The Suffolk coastline is a landscape of significant ecological, recreational and historical value.

Several areas along the Suffolk coastline are at risk of flooding from the sea and from rivers. One at-risk area lies behind the shingle ridge of Minsmere frontage.

Minsmere frontage stretches from Minsmere Cliffs in the north to the Sizewell power station in the south. The flood risk area is shown on Figure 1. It stretches inland from the coast to Middleton, along the Minsmere River valley, and to Leiston Common. The area supports a mix of habitats including dry heaths, freshwater reedbeds, grazing marsh and river banks, as well as shingle banks on the beach and dunes that support plant life (vegetated shingle).

Some of these habitats and the species they support are of international importance. As such they are protected under a number of conservation listings, including being designated as a Site of Special Scientific Interest (SSSI). The area also includes the RSPB Minsmere nature reserve and Sizewell Marshes SSSI.



Photo 1: Aerial view of Minsmere RSPB reserve



Figure 1: The study area



Photo 2: Vegetated shingle banks at Minsmere

The area is of considerable archaeological interest and Leiston Abbey, a Scheduled Monument, is situated within the site. Minsmere also falls within the Suffolk Coast and Heaths Area of Outstanding Natural Beauty because of its landscape features.

Minsmere is important in terms of tourism and recreation, supporting activities such as walking, birdwatching and fishing - all of which all contribute to the local economy.

Under nature conservation legislation we are legally required to manage flood risk to protect the habitat and species of the site. To assess how we should do this, we need to consider the likely changes to the site through coastal processes and understand how this will affect the way we manage flood risk in the future.

The purpose of this document is to focus on immediate flood risk management issues at Minsmere.

The present site

Minsmere is protected from coastal flooding by two types of defence.

The primary defence is the line of natural and modified sand dunes and shingle ridge. The secondary defence is a clay embankment along the back of the dunes at the northern part of the frontage.

Along with the current coastal flood defences, flood management is dependent on a network of ditches. These are drained via gravity by a sluice - a channel that carries off excess water. The sluice outfall is midway along the coastal frontage and flows into the sea.

Part of the site is managed by the RSPB, who maintain suitable water levels for wildlife. We have worked in the past with the RSPB, National Trust and Natural England to maintain the present line of coastal defence and the sluice outfall.

The coastal defences at the northern end of the site are under significant pressure from erosion by the sea, which is threatening their long-term stability.

In addition, climate change is causing sea levels to rise. This will increase erosion on the frontage and therefore increases the risk that the defences will be breached in the future.

We are already seeing the effects of this. In recent years the sand and shingle ridge at the northern end of the site has been breached on a number of occasions, and the ridge has rolled back onto the secondary defence. These events were caused by tidal surges, which led to flooding of the area between the dunes and secondary clay embankment in November 2006 and 2007.

As a consequence, the sluice outfall was blocked with shingle, triggering freshwater flooding of the site. At the same time seawater also entered the site close to the sluice, due to problems with one of the sluice gates.

Over the last decade there have also been many instances of flooding of parts of the RSPB reserve following heavy rainfall.

The flood risk predominantly applies to wildlife habitats and species. The surrounding settlements lie on higher ground on the edge of the floodplain and are only likely to be at risk from extreme flooding.

Future change

To understand how to manage flood risk we need to understand how Minsmere is likely to evolve.

We have carried out a number of studies to predict any changes likely to occur at the Minsmere site if no action was taken to manage flood risk. We know that the primary coastal defences to the north of the sluice currently provide the lowest level of protection, and that future erosion is most likely to cause the defences north of Coney Hill to breach. The arrow shown on Photo 3 indicates where the defences are under greatest pressure.

Should breaching occur, the area behind will be flooded with seawater on an increasingly regular basis. Without further action, this is likely to happen within the next 20 years.

The defences south of the sluice are higher and less likely to be at risk from coastal erosion over the next 100 years. The soft cliffs to the north are eroding landward while the flood defences are fixed. This means that there is an increasing risk of floodwater outflanking the defences where the embankment and cliffs meet.

The tidal sluice outfall extends from the beach into the sea. This structure helps to stabilise the shoreline. Failure of this structure may affect local coastal processes, leading to potential erosion of the shoreline around the sluice and to the south.

Climate change and sea level rise will put increasing pressure on the existing defences and will also reduce the ability of the sluice and outfall to drain by gravity.

As the effectiveness of the sluice and outfall declines, freshwater flooding of the land behind the defences will become more frequent.



Photo 3: The Minsmere Frontage and Area of Increased Risk of Breach

The need for the study

The coast and wetlands at Minsmere represent some of the most important natural habitats in the UK. These habitats are under threat and are likely to change as a result of coastal change and sea level rise.

Under nature conservation legislation we are legally obliged to manage flood risk for the internationally designated habitats and species at Minsmere.

Pressure on the shoreline presents a risk to the Special Area of Conservation (SAC) shingle habitat at the northern end of the site. Coupled with this is the increasing risk of seawater flooding and erosion of habitat that supports the species of bird, plants and invertebrates that are protected as part of the Special Protection Area and Ramsar conservation designations.

Recently, coastal surge events and periods of intense and sustained rainfall have also led to freshwater flooding of the site because of the limited drainage capacity of the sluice. This has an impact on the bittern population which relies on stable freshwater water levels within the extensive area of reedbeds at Minsmere.

Flood management measures will focus on reducing the significant risks to protected species and habitats in the area.

We aim to provide a short-term solution to the immediate problem of increased tidal flood risk to the north of the site and to consider issues associated with freshwater drainage.

Longer term issues associated with the coastal frontage, the tidal sluice outfall and freshwater drainage will be addressed in future studies.



Photo 4: Aerial view of the reedbed habitats at Minsmere

Project objectives

Our aim is to develop a sustainable solution which considers the needs of all local interests while meeting our legal duties.

Nature Conservation

In line with Government policy, we aim to protect habitats and species within internationally designated sites where it is sustainable to do so. However, with rising sea levels, the building of new defences may be technically difficult and expensive on dynamic coastlines. Where it is not feasible to maintain sites and where designated habitats or species are at risk of being lost due to coastal change, then we aim to replace these in more sustainable locations elsewhere.

We will be flexible in our approach to this issue. We will identify a short-term solution that protects as much of the freshwater habitats as is sustainable, but will allow for future management changes as the habitats develop over the long term.

Risks to people

The flood risk to people in the area is low. There are a few properties on the edge of the floodplain, the majority of which are only at risk from extreme events. We will continue to manage the risk to people and ensure that any works from this project do not increase the flood risk to people and property.

Land Use

We will consider how flood risk management affects the surrounding land use, now and in the future. At Minsmere this means considering the interests of local landowners and the Sizewell power stations. We aim to find a solution that is acceptable to all parties involved.

The options

We thoroughly examined potential options with the aim of selecting the most effective and sustainable approach to deal with the immediate risk from increased tidal flooding in the north of the site.



Photo 5: The early morning beach with Sizewell on the horizon

We have tested the sustainability of a number of potential options using four important questions, as shown in Table 1. We considered the effects of these options on the protected species and habitats of Minsmere and identified whether the options are working with, rather than against, coastal processes. We also looked at whether the option is technically feasible and does not entail excessive cost.

Table 1. Sustainability appraisal criteria

1. Will it adversely affect the international nature conservation features of the site?
2. Does it work with rather than against coastal processes?
3. Is it technically feasible?
4. Will it require excessive capital or maintenance costs disproportionate to the importance of the feature under threat?

The following summarises the options we have considered and their likely effects.

Do the minimum

Continue existing management regime

Under this approach we would carry out small scale works to repair minor breaches in the defences. We would also address health and safety concerns if damage occurs. There would be no capital investment in the defences, and the standard of protection will be reduced as sea levels rise. The defences would breach with increasing frequency and severity over time, until eventually repairing them would become unsustainable.

Repairing the defences will contribute to the continued protection of the freshwater habitats. However, when maintenance is no longer viable, breach events will adversely affect these habitats. Works to repair the defences may also lead to damage to the protected vegetation on the shingle banks and beach. Although this low cost option is potentially suitable for the very short term, it does not work with the natural coastal processes and will not provide a longer term solution. Therefore this option does **not meet the project objectives.**

Hold the existing line

Take action to maintain the defences in their current position

This approach would involve the construction of new structures to halt the erosion of the defences. The options considered were:

- **Beach recharge:** the addition of sediment to build up the height and width of the shingle beach. This option would protect the existing freshwater habitats by limiting the need for land-based construction, keeping large scale disturbance to a minimum. However, every time this work is carried out we would damage the protected shingle bank vegetation. Recharge events will be required every five years and re-profiling of the beach may be needed every year, leading to high capital and maintenance costs and repeated damage to the designated habitat. **This option does not meet the project objectives.**
- **Rock armour groynes:** construction of a series of rock groynes (banks) to reduce sediment movement and encourage retention of the beach. Regular recharge operations would be needed. Under this option there



Photo 6: Aerial view of the beach and landward freshwater habitat

would be no loss of freshwater habitat because the defence line would be held. However, the shingle vegetation will be damaged during construction and maintenance of the groynes and the necessary beach recharging. The option is also costly and will work against existing coastal processes, therefore **this option does not meet the project objectives.**

- **Secondary Defences:** strengthening, raising and armouring the secondary line of defence north of Minsmere Sluice. Construction of bunds (embankments) would tie in the defences to high ground. A small loss of reedbed will be inevitable because the defence will need to be widened, but overall it is considered that the loss of protected freshwater habitat will be smaller under this option in comparison to the others. However, future squeeze of the shoreline may adversely affect the shingle habitat, especially in locations where erosion pressure is high. This option is costly and does not allow the coastline to evolve naturally. **It does not meet the project objectives.**

Retreat the existing line

Allow the defences to retreat naturally to a new landward defence line.

Two potential options were considered:

- **Full managed realignment:** the relocation of the existing defence line to higher ground surrounding the RSPB reserve, followed by withdrawal of maintenance of the primary and secondary defences. Additional flood barriers would be constructed to protect assets landward of the RSPB reserve and the sluice would be replaced with separate outlets for each river in the area. Realignment of the defence would allow the front line to roll back naturally. Protected freshwater habitats in front of the new alignment would be lost in time, and these would need to be recreated elsewhere. This loss, along with the construction of the new defence, will cause significant disturbance to the site and incur very large costs. **This option is not deemed necessary and was therefore rejected.**
- **Partial managed realignment:** maintenance would be withdrawn from the primary defence, but the secondary defences would be maintained. The existing embankment at Coney Hill would be improved to protect the vast majority of the freshwater wetland area from flooding. Any minor damage to the secondary defences north of Coney Hill may be repaired, but these are expected to be breached within the next 20 years, leading to occasional tidal flooding of freshwater habitats in this area. The freshwater habitat would therefore need to be recreated elsewhere. However, this regime will allow natural coastal processes to continue and the shingle bank would roll back gradually, to the benefit of the shingle vegetation. **On balance, this option best meets the project objectives.**

Our solution

We have identified a sustainable solution to manage the immediate issue of tidal flood risk at Minsmere site.

We believe that partial managed realignment of the defences in the north of the site will provide the most sustainable solution to the management of coastal flood risk at Minsmere in the short term. We will allow the banks at the northern end of the site to breach and the vegetated shingle habitat to move inland under natural processes.

We will make improvements to Coney Hill cross bank (see Figure 1) to isolate the area to the north known as North Marsh from any flooding from the sea. We will also make minor improvements to the secondary defence and the sluice to ensure that the site is protected against flood risk in the short term.

The freshwater reedbed habitat within North Marsh will change and we will find replacement habitat for freshwater areas that are lost.

The conclusions of this project will be considered in the review of the Lowestoft to Harwich Shoreline Management Plan (SMP). Further studies will be carried out which will take into account the longer term recommendations of the SMP. These studies will address the longer term issues at the site associated with freshwater drainage, long-term coastal flood risk and habitat management.

To ensure that our solution effectively manages flood risk over the short term, works that are required in the near future need to be identified now and put into action.

We have formulated a plan of work, as illustrated on Figure 2. We will:

- Strengthen and raise the existing Coney Hill cross bank. This will contain the effects of flooding in the north of the site and protect freshwater habitats south of the bank.
- Construct two bunds connecting the primary and secondary defences just south of Coney Hill cross bank. This will control the drainage of overspill water and help ensure the stability of the defence.
- Identify and secure suitable replacement habitat for the eventual loss of freshwater habitat at North Marsh.
- Repair any breaches to the secondary defences if considered necessary in the short term. Various factors will be considered when assessing the need for repair, including the extent of the damage, the timing of the breach, the presence of protected habitats and species, and the status of replacement habitat.
- Monitor the condition of the defences and the sluice and the continued effects of coastal processes. This will help inform future decisions regarding management of the site.

We will continue to liaise with the Steering Group (who represent organisations and the interests of local parties and conservation) and local interest groups to identify further flood risk management opportunities as they arise.



Photo 7: Minsmere reserve with North Marsh in the foreground

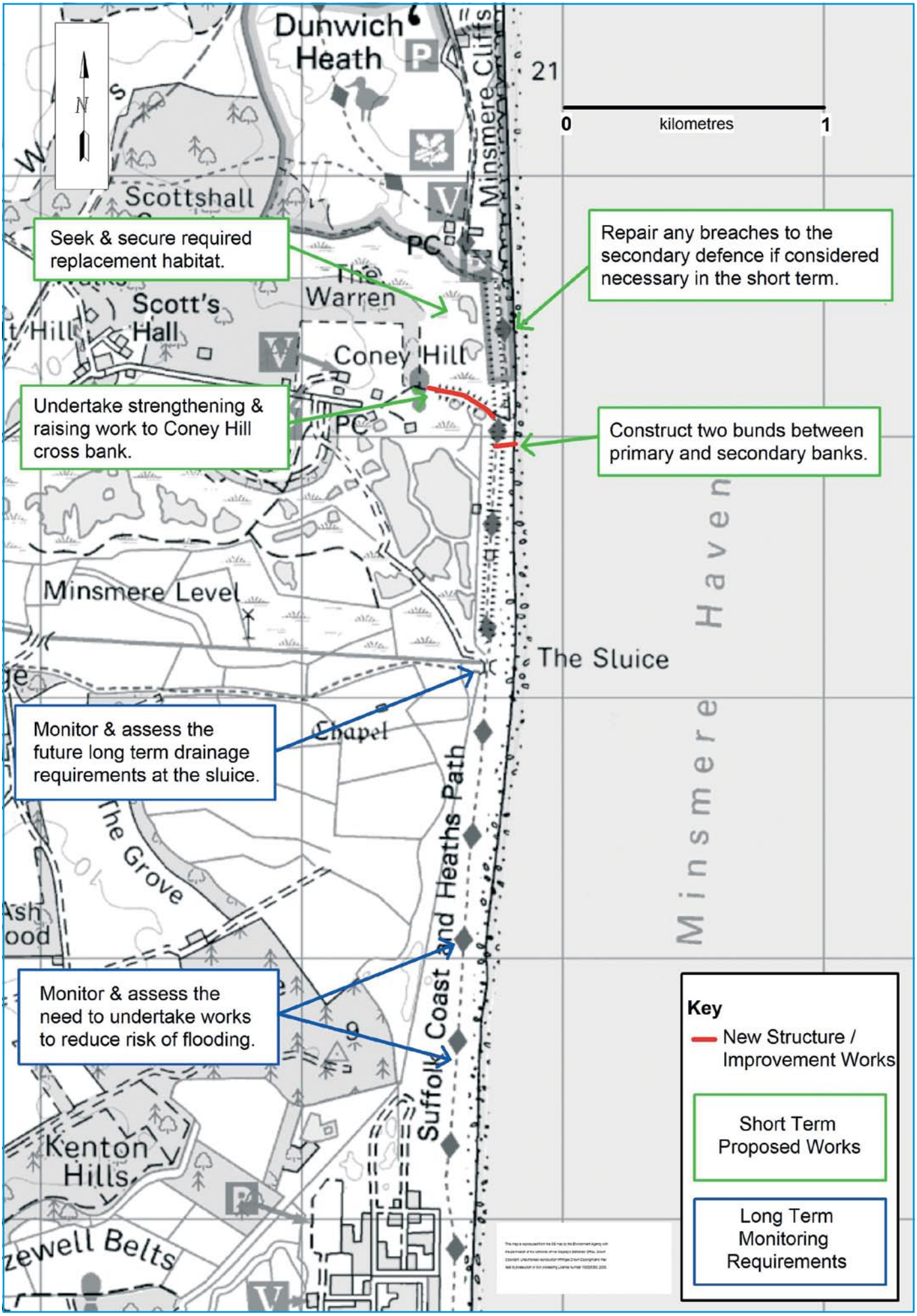


Figure 2: Proposed works

What happens next?

We would like to hear your views on the proposed works shown on Figure 2.

You can contact us in writing or by e-mail - please see page xx for contact details.

After we've considered the responses received from this consultation, the next stages of the study are to:

- Complete the initial design of the short-term work.
- Undertake environmental assessment of the short-term work.
- Prepare a business case for the funding of construction works. This is subject to approval at national level by the Environment Agency.

These and future stages of the project are summarised in Table 2.

During the next stages of the project we will continue to work closely with the Steering Group. We will also consult with other statutory bodies, landowners, residents and any other interested parties who express an interest in response to this consultation.

The future progression of the study, and most importantly the implementation of the most sustainable solution, will depend on the success of our applications for funding, planning approval and other consents.

Table 2. Key future stages of study

Stage	Key Tasks	Formal Output	Timescale
Appraisal and scoping	Review of consultation feedback	Response letters and meetings as required	February 2009
	Completion of the initial design of the preferred works	Preparation and issue of initial design drawing and other reports	Spring 2009
	Environmental scoping of short term works	Preparation and issue of Scoping Report	Spring 2009
	Preparation of business case for the short term work	Preparation and issue of Project Appraisal Report	Spring 2009
Application for funding approval			
Detailed design and assessment (subject to funding approval)	Development of detailed design drawing	Preparation and issue of detailed design	Summer 2009 – Spring 2010
	Environmental impact assessment of preferred option	Preparation and issue of Environmental Statement (if required)	Summer 2009 – Spring 2010
Applications for planning permission and other consents (as required)			Winter 2009 – Summer 2010

Consultees

The following organisations are being consulted as part of this consultation process.

Aldeburgh and District Angling Club

British Association for Shooting and Conservation Eastern England Centre

British Energy

British Nuclear Group

British Trust for Ornithology

Cliff House Holiday Park

Crown Estates

Defra

DCLG

Dunwich Parish Council

Dunwich Private Caravan Park

East Anglia Fishermen's Association

East of England Development Agency

Eastern Sea Fisheries Joint Committee

English Heritage

Essex and Suffolk Water

Leiston Parish Council

Marine and Fisheries Agency

Middleton Parish Council

Ministry of Defence

Minsmere Levels Stakeholder Group

National Trust

Natural England

Norfolk & Suffolk Anglers Consultative Association

Ramblers Association

Royal Yachting Association

RSPB

Sizewell Shoreline Management Group

Suffolk Biological Records Centre

Suffolk Coast and Heaths Project

Suffolk Coastal District Council

Suffolk County Anglers Association

Suffolk County Council

Suffolk Wildlife Trust

Theberton and Eastbridge Parish Council

Trinity House Lighthouse Service

Westleton Parish Council

Wildfowl and Wetlands Trust

Contacts

We welcome your comments on the proposed works at Minsmere. We will take account of all comments received by Friday 13 February 2009 before finalising our proposals.

We would be grateful for your comments on the proposed preferred option. You can write to us:

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or email:

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Environment Bill

[AS AMENDED IN PUBLIC BILL COMMITTEE]

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(1) The Secretary of State may by regulations make provision for and in relation to a register of biodiversity gain sites ("the biodiversity gain site register").

(2) A biodiversity gain site is land where—

(a) a person is required under a conservation covenant or planning obligation to carry out works for the purpose of habitat enhancement,

(b) that or another person is required to maintain the enhancement for at least 30 years after the completion of those works, and

(c) for the purposes of Schedule 7A to the Town and Country Planning Act 1990 the enhancement is made available to be allocated (conditionally or unconditionally, and whether for consideration or otherwise) in accordance with the terms of the covenant or obligation to one or more developments for which planning permission is granted.

(3) Regulations under this section must provide for the information in the register to be accessible to members of the public.

(4) Regulations under this section may in particular make provision about—

(a) the person who is to establish and maintain the biodiversity gain site register (who may be the Secretary of State, Natural England or another person);

(b) circumstances in which land is or is not eligible to be registered;

(c) applications to register land in the register;

(d) the information to be recorded in relation to any land that is registered;

(e) amendments to the register;

(f) removal of land from the register;

(g) fees payable in respect of any application under the regulations.

(5) Provision under subsection (4)(c) may in particular include provision about—

(a) who is entitled to apply to register land in the biodiversity gain site register;

(b) the procedure to be followed in making an application;

(c) the information to be provided in respect of an application;

(d) how an application is to be determined;

(e) appeals against the rejection of an application;

(f) financial penalties for the supply of false or misleading information in connection with an application.

(6) Provision under subsection (4)(d) may in particular require the recording of the following in relation to any land registered in the biodiversity gain site register—

(a) the location and area of the land;

(b) the works to be carried out on the land and the habitat enhancement to be achieved by them;

(c) information about the habitat of the land before the commencement of those works;

(d) the person who applied to register the land and (if different) the person by whom the requirement to carry out the works or maintain the habitat enhancement is enforceable;

(e) any development to which any of the habitat enhancement has been allocated;

(f) the biodiversity value (for the purposes of Schedule 7A to the Town and Country Planning Act 1990) of any such habitat enhancement in relation to any such development.

(7) Regulations under this section making provision under subsection (4)(g) or (5)(f) are subject to the affirmative procedure.

(8) Other regulations under this section are subject to the negative procedure.

(9) In this section "development", "habitat enhancement", "planning obligation" and "planning permission" have the same meanings as in Schedule 7A to the Town and Country Planning Act 1990.

94 Biodiversity credits

(1) The Secretary of State may make arrangements under which a person who is entitled to carry out the development of any land may purchase a credit from the Secretary of State for the purpose of meeting the biodiversity gain objective referred to in Schedule 7A to the Town and Country Planning Act 1990.

(2) A credit is to be regarded for the purposes of that Schedule as having such biodiversity value as is determined under the arrangements.

(3) The arrangements may in particular include arrangements relating to—

(a) applications to purchase credits;

(b) the amount payable in respect of a credit of a given value;

(c) proof of purchase;

(d) reimbursement for credits purchased for development which is not carried out.

(4) In determining the amount payable under the arrangements for a credit of a given value the Secretary of State must have regard to the need to determine an amount which does not discourage the registration of land in the biodiversity gain sites register.

(5) The Secretary of State must publish information about the arrangements, including in particular the amount payable for credits.

(6) The Secretary of State may use payments received under arrangements under this section for the following purposes (only)—

(a) carrying out works, or securing the carrying out of works, for the purpose of habitat enhancement (within the meaning of Part 7A of the Town and Country Planning Act 1990) on land in England;

(b) purchasing interests in land in England with a view to carrying out works, or securing the carrying out of works, for that purpose;

(c) operating or administering the arrangements.

(7) The references to works in subsection (6) do not include works which the Secretary of State is required to carry out apart from this section by virtue of any enactment.

(8) The Secretary of State must publish reports relating to the discharge of the Secretary of State's functions under subsections (1) and (6).

(9) A report must relate to a period not exceeding a year which—

(a) in the case of the first report, begins on the date on which Schedule 7A to the Town and Country Planning Act 1990 comes into force in relation to any development (within the meaning of Part 3 of that Act), and

(b) in the case of any subsequent report, begins on the day after the last day of the period to which the previous report related.

(10) A report must set out—

(a) the total payments received under arrangements under this section in the period to which the report relates,

(b) how those payments have been used, and

(c) where those payments have been used for the purpose of carrying out or securing the carrying out of works for the purpose of habitat enhancement, the projected biodiversity value of the habitat enhancement at such time or times after completion of the works as the Secretary of State considers it appropriate to specify.

Biodiversity objective and reporting

95 General duty to conserve and enhance biodiversity

(1) Section 40 of the Natural Environment and Rural Communities Act 2006 (duty to conserve biodiversity) is amended in accordance with subsections (2) to (7).

(2) In the heading, after "conserve" insert "and enhance".

(3) For subsections (A1) and (1) substitute—

"(A1) For the purposes of this section "the general biodiversity objective" is the conservation and enhancement of biodiversity in England through the exercise of functions in relation to England.

(1) A public authority which has any functions exercisable in relation to England must from time to time consider what action the authority can properly take, consistently with the proper exercise of its functions, to further the general biodiversity objective.

(1A) After that consideration the authority must (unless it concludes there is no new action it can properly take)—

(a) determine such policies and specific objectives as it considers appropriate for taking action to further the general biodiversity objective, and

(b) take such action as it considers appropriate, in the light of those policies and objectives, to further that objective.

(1B) The requirements of subsection (1A)(a) may be satisfied (to any extent) by revising any existing policies and specific objectives for taking action to further the general biodiversity objective.

(1C) The first consideration required by subsection (1) must be completed by the authority within the period of one year beginning with the day on which section 95 of the Environment Act 2020 comes into force.

(1D) Any subsequent consideration required by subsection (1) must be completed no more than five years after the completion of the authority's previous consideration.

(1E) A determination required by subsection (1A)(a) must be made as soon as practicable after the completion of the consideration to which it relates.

(1F) Nothing in this section prevents the authority from—

(a) determining or revising policies and specific objectives at any time, or

(b) taking action to further the general biodiversity objective at any time."

(4) In subsection (2) for "subsection (1)" substitute "subsections (1) and (1A)".

(5) After subsection (2) insert—

"(2A) In complying with subsections (1) and (1A) the authority must in particular have regard to—

(a) any relevant local nature recovery strategy, and

(b) any relevant species conservation strategy or protected site strategy prepared by Natural England."

(6) For subsection (3) substitute—

"(3) The action which may be taken by the authority to further the general biodiversity objective includes, in particular, action taken for the purpose of—

(a) conserving, restoring or otherwise enhancing a population of a particular species, and

(b) conserving, restoring or otherwise enhancing a particular type of habitat."

(7) After subsection (5) insert—

"(6) This section has effect in relation to Her Majesty's Revenue and Customs with the following modifications—

(a) the omission from subsection (A1) of the words "in England" and "in relation to England";

(b) the omission from subsection (1) of the words from "which" to "England".

(7) In this section references to England include the territorial sea adjacent to England."

(8) In section 41 of that Act (biodiversity lists and action (England))—

(a) in subsection (1), after "conserving" insert "or enhancing";

(b) in subsection (3) for "and (2)" substitute "and (1A)".

96 Biodiversity reports

(1) After section 40 of the Natural Environment and Rural Communities Act 2006 insert—

"40A Biodiversity reports

(1) This section applies to—
  (a) a local authority in England other than a parish council,
  (b) a local planning authority in England, and
  (c) a designated authority (see subsection (8)(a)).

(2) A public authority to which this section applies ("the authority") must publish biodiversity reports in accordance with this section.

(3) A biodiversity report so published must contain—
  (a) a summary of the action which the authority has taken over the period covered by the report for the purpose of complying with its duties under section 40(1) and (1A),
  (b) a summary of the authority's plans for complying with those duties over the period of five years following the period covered by the report,
  (c) any quantitative data required to be included in the report by regulations under subsection (8)(b), and
  (d) any other information that the authority considers it appropriate to include in the report.

(4) If the authority is a local planning authority, its biodiversity report must also contain—
  (a) a summary of the action taken by the authority in carrying out its functions under Schedule 7A to the Town and Country Planning Act 1990 (biodiversity gain as condition of planning permission) over the period covered by the report,
  (b) information about any biodiversity gains resulting or expected to result from biodiversity gain plans approved by the authority during that period, and
  (c) a summary of the authority's plans for carrying out those functions over the five year period following the period covered by the report.

(5) A biodiversity report—
  (a) must specify the period covered by the report, and
  (b) must be published within the period of 12 weeks following the last day of that period.

(6) The authority's first biodiversity report must cover a period chosen by the authority which—
  (a) is no longer than three years, and
  (b) begins with the day on which the authority first becomes subject to the duty under subsection (2).

(7) A subsequent biodiversity report made by the authority must cover a period chosen by the authority which—
  (a) is no longer than five years, and

(b) begins with the day after the last day of the period covered by its most recent biodiversity report.

(8) The Secretary of State may by regulations—

(a) provide for specified public authorities, or public authorities of a specified description, to be designated authorities for the purposes of this section;

(b) require biodiversity reports to include specified quantitative data relating to biodiversity in any area of land in England in relation to which the authority exercises any functions.

In this subsection "specified" means specified in the regulations.

(9) Public authorities with no functions exercisable in relation to England may not be designated under subsection (8)(a).

(10) The power to make regulations under subsection (8) is exercisable by statutory instrument.

(11) A statutory instrument containing regulations under subsection (8) is subject to annulment in pursuance of a resolution of either House of Parliament.

(12) Terms used in this section and section 40 have the same meaning as in that section".

Local nature recovery strategies

97 Local nature recovery strategies for England

(1) There are to be local nature recovery strategies for areas in England.

(2) Together the local nature recovery strategies are to cover the whole of England.

(3) The Secretary of State is to determine the areas within England to which individual local nature recovery strategies are to relate.

(4) The area of a local authority, other than a county council, may not be split between local nature recovery strategies.

(5) Section 40(2A) of the Natural Environment and Rural Communities Act 2006 (duty to conserve biodiversity) makes provision about the duties of public authorities in relation to local nature recovery strategies.

98 Preparation of local nature recovery strategies

(1) A local nature recovery strategy for an area ("the strategy area") is to be prepared and published by the responsible authority.

(2) The responsible authority for a local nature recovery strategy is such one of the following authorities as is appointed by the Secretary of State—

(a) a local authority whose area is, or is within, the strategy area;

(b) the Mayor of London;

(c) the mayor for the area of a combined authority established under section 103 of the Local Democracy, Economic Development and Construction Act 2009;

(d) a National Park authority in England;

(3) Regulation under sub-paragraph (2) must contain provision for ensuring that the power in question is exercised by a person only where the person reasonably believes there has been a failure to comply with a requirement of regulations under this Schedule.

Civil sanctions

9 (1) The regulations may make provision for, about or connected with the imposition of civil sanctions in respect of failures to comply with the regulations, or the obstruction of or failure to assist a person on whom functions are conferred by the regulations.

(2) The regulations may make provision for appeals against such sanctions.

(3) In this paragraph "civil sanction" means a sanction of a kind for which provision may be made under Part 3 of the Regulatory Enforcement and Sanctions Act 2008 (fixed monetary penalties, discretionary requirements, stop notices and enforcement undertakings).

(4) The regulations may include provision for the imposition of sanctions of that kind whether or not—

(a) the conduct in respect of which the sanction is imposed constitutes an offence,

(b) the enforcement authority is a regulator for the purposes of Part 3 of the Regulatory Enforcement and Sanctions Act 2008, or

(c) the relevant national authority may make provision for the imposition of sanctions under that Part.

SCHEDULE 10

Section 65

ENFORCEMENT POWERS

Powers to search and seize vehicles in connection with waste offences

1 In section 5(6) of the Control of Pollution (Amendment) Act 1989 (constable's power to seize vehicles and contents)—

(a) in paragraph (b) after "presence of" insert "or at the request of";

(b) in paragraph (c) for "without such an officer present" substitute "in any other case".

2 In section 34B(6) of the Environmental Protection Act 1990 (constable's power to seize vehicles and contents)—

(a) in paragraph (b) after "presence of" insert "or at the request of";

(b) in paragraph (c) for "without such an officer present" substitute "in any other case".

Powers of direction in relation to waste

3 (1) Section 57 of the Environmental Protection Act 1990 (power to give directions) is amended as follows.

(2) In subsection (2) omit the words from "with a view" to the end.

(3) After subsection (2) insert—

"(2A) The appropriate Minister may, by notice in writing—

(a) direct a registered waste carrier to collect waste which is being kept on specified land and deliver it to a specified person on specified terms;

(b) direct any person who—

(i) is keeping waste on any land, or

(ii) owns or occupies land on which waste is being kept,

to facilitate collection of the waste by a specified registered waste carrier to whom a direction in respect of the waste is given under paragraph (a)."

(4) In subsection (4), for "of treating or disposing of" substitute "in relation to".

(5) After subsection (4) insert—

"(4A) A direction under subsection (2A)(b) may require the person to whom it is given—

(a) to pay to the specified registered waste carrier the reasonable costs of collecting and delivering the waste;

(b) to pay to the specified person to whom the waste is delivered ("P") the reasonable costs incurred by P in relation to the waste (including any costs P is required by a direction under this section to pay to another person)."

(6) In subsection (7) for the words from ", where" to the end substitute "pay any costs mentioned in subsection (4)."

(7) After subsection (7) insert—

"(7A) The appropriate Minister may pay any costs mentioned in subsection (4A)."

(8) In subsection (8), before the definition of "specified" insert—

""appropriate Minister" means—

(a) the Secretary of State, in relation to waste being kept on land in England, and

(b) the Welsh Ministers, in relation to waste being kept on land in Wales;

"registered waste carrier" means a person registered under the Control of Pollution (Amendment) Act 1989 as a carrier of controlled waste;".

Powers of entry in relation to pollution control etc

4 The Environment Act 1995 is amended as follows.

5 (1) Section 108 (powers of enforcing authorities and their authorised officers) is amended as follows.

(2) In subsection (4), after paragraph (k) insert—

"(ka) as regards any premises which an English or Welsh authorised person has power to enter by virtue of paragraph (a), for the purposes of an examination or investigation under paragraph (c)—

(i) to search the premises;

(ii) to seize and remove documents or anything else found on the premises (other than an article or substance within paragraph (g));

(iii) to require any information which is stored in electronic form and is accessible from the premises to be produced in a form in which it can be removed and—

(a) in which it is visible and legible, or

(b) from which it can readily be produced in a visible and legible form;

(iv) to operate any equipment found on the premises for the purposes of producing such information in such a form;".

(3) In subsection (6), omit paragraph (a).

(4) After subsection (7) insert—

"(7A) An English or Welsh authorised person may not exercise the powers in subsection (4)(ka) without the authority of a warrant by virtue of Schedule 18 to this Act.

This is subject to subsections (7B) and (7C).

(7B) An English or Welsh authorised person may exercise a power in subsection (4)(ka)(ii) to (ka)(iv) in relation to a thing without the authority of a warrant if the person has reasonable grounds for believing that —

(a) it is evidence of a failure to comply with any provision of the pollution control enactments or flood risk activity enactments, and

(b) exercising the power is necessary to prevent it being concealed, lost, altered or destroyed.

(7C) Subsection (7A) does not require the authority of a warrant for doing something within the powers in subsection (4)(ka) if, and so far as, it may be done in exercise of another power conferred by subsection (4).

(7D) Where anything seized or removed from premises under subsection (4)(ka) contains protected material, that material—

(a) may not be used for the purposes of an examination or investigation under subsection (4)(c), and

(b) must be returned to the premises from which it was removed, or to the person who had possession or control of it immediately before it was removed, as soon as reasonably practicable after it is identified as protected material.

(7E) Subsection (7D) does not prevent any part of a thing containing protected material which is not protected material being used for the purposes of an examination or investigation, retained or copied.

(7F) "Protected material" means—

(a) material subject to legal professional privilege,

(b) excluded material within the meaning of section 11 of the Police and Criminal Evidence Act 1984, or

(c) journalistic material, within the meaning of section 13 of that Act, which is not excluded material."

(5) After subsection (12) insert—

"(12A) Subject to subsection (7D), anything seized or removed under subsection (4)(ka) may be retained for so long as is necessary in all the circumstances."

(6) In subsection (15)—

(a) after the definition of "authorised person" insert—

""document" includes anything in which information of any description is recorded (by any means) and any part of such a thing;";

(b) after the definition of "enforcing authority" insert—

""English or Welsh authorised person" means a person authorised under subsection (1) or (2) by the Secretary of State, the Welsh Ministers, the Agency, the Natural Resources Body for Wales, a waste collection authority or a local enforcing authority in England or Wales;";

(c) in the definition of "pollution control functions" in relation to a waste collection authority, in paragraph (a) after "46" insert "to 46D".

6 (1) Schedule 18 (supplemental provision about powers of entry) is amended as follows.

(2) In paragraph 2—

(a) after sub-paragraph (2) insert—

"(2A) A justice of the peace may by warrant authorise an English or Welsh authorised person, designated for the purpose by the person who authorised them, to exercise the powers in section 108(4)(ka) in accordance with the warrant and, if need be, by force.

(2B) The justice may do so only if satisfied that there are reasonable grounds for believing that—

(a) there is material on or accessible from the premises in question which is likely to be of substantial value (by itself or together with other material) to an examination or investigation under section 108(4)(c), and

(b) it is impracticable to communicate with a person entitled to grant access to it, or access to it is unlikely to be granted unless a warrant is produced.";

(b) omit sub-paragraph (3).

(3) In paragraph 3 after "shall" insert ", if so required,".

"authorised member of the CMA"—

(a) in relation to a power exercisable in connection with an appeal in respect of which a group has been constituted by the chair of the CMA under Schedule 4 to the Enterprise and Regulatory Reform Act 2013, means a member of that group who has been authorised by the chair of the CMA to exercise that power;

(b) in relation to a power exercisable in connection with an application for permission to bring an appeal, or otherwise in connection with an appeal in respect of which a group has not been so constituted by the chair of the CMA, means—

(i) any member of the CMA Board who is also a member of the CMA panel, or

(ii) any member of the CMA panel authorised by the Secretary of State (whether generally or specifically) to exercise the power in question;

"CMA Board" and "CMA panel" have the same meaning as in Schedule 4 to the Enterprise and Regulatory Reform Act 2013;

"statement of truth", in relation to the production of a statement or provision of information by a person, means a statement that the person believes the facts stated in the statement or information to be true;

"working day" means any day other than—

(a) Saturday or Sunday;

(b) Christmas Day or Good Friday;

(c) a day which is a bank holiday in England and Wales under the Banking and Financial Dealings Act 1971.

(2) References in this Schedule to a party to an appeal are references to—

(a) the appellant, or

(b) the Authority."

SCHEDULE 14

Section 92

BIODIVERSITY GAIN AS CONDITION OF PLANNING PERMISSION

PART 1

BIODIVERSITY GAIN CONDITION

1 In the Town and Country Planning Act 1990, after section 90 insert—

"Biodiversity gain

90A Biodiversity gain in England

Schedule 7A (biodiversity gain in England) has effect."

2 In that Act, after Schedule 7 insert—

"SCHEDULE 7A

BIODIVERSITY GAIN IN ENGLAND

PART 1

OVERVIEW AND INTERPRETATION

Overview

1 (1) This Schedule makes provision for grants of planning permission in England to be subject to a condition to secure that the biodiversity gain objective is met.

(2) Paragraphs 2 to 12 have effect for the purposes of this Schedule.

Biodiversity gain objective

2 (1) The biodiversity gain objective is met in relation to development for which planning permission is granted if the biodiversity value attributable to the development exceeds the pre-development biodiversity value of the onsite habitat by at least the relevant percentage.

(2) The biodiversity value attributable to the development is the total of—

(a) the post-development biodiversity value of the onsite habitat,

(b) the biodiversity value, in relation to the development, of any registered offsite biodiversity gain allocated to the development, and

(c) the biodiversity value of any biodiversity credits purchased for the development.

(3) The relevant percentage is 10%.

(4) The Secretary of State may by regulations amend this paragraph so as to change the relevant percentage.

Biodiversity value and the biodiversity metric

3 References to the biodiversity value of any habitat or habitat enhancement are to its value as calculated in accordance with the biodiversity metric.

4 (1) The biodiversity metric is a document for measuring, for the purposes of this Schedule, the biodiversity value or relative biodiversity value of habitat or habitat enhancement.

(2) The biodiversity metric is to be produced and published by the Secretary of State.

(3) The Secretary of State may from time to time revise and republish the biodiversity metric.

(4) Before publishing or republishing the biodiversity metric the Secretary of State must consult such persons as the Secretary of State considers appropriate.

(5) The Secretary of State may by regulations make transitional provision in relation to the revision and republication of the biodiversity metric.

Pre-development biodiversity value

5 (1) In relation to any development for which planning permission is granted, the pre-development biodiversity value of the onsite habitat is the biodiversity value of the onsite habitat on the relevant date.

(2) The relevant date is—

(a) in a case in which planning permission is granted on application, the date of the application, and

(b) in any other case, the date on which the planning permission is granted.

(3) But the person submitting the biodiversity gain plan for approval and the planning authority may agree that the relevant date is to be a date earlier than that specified in sub-paragraph (2)(a) or (b) (but not a date which is before the day on which this Schedule comes into force in relation to the development).

(4) This paragraph is subject to paragraphs 6 and 7.

6 If—

(a) a person carries on activities on land on or after 30 January 2020 otherwise than in accordance with—

(i) planning permission, or

(ii) any other permission of a kind specified by the Secretary of State by regulations, and

(b) as a result of the activities the biodiversity value of the onsite habitat referred to in paragraph 5(1) is lower on the relevant date than it would otherwise have been,

the pre-development biodiversity value of the onsite habitat is to be taken to be its biodiversity value immediately before the carrying on of the activities.

7 Where planning permission is granted in respect of land which is registered in the biodiversity gain site register under section 93 of the Environment Act 2020, the pre-development biodiversity value of the land is the total of—

(a) the biodiversity value of the onsite habitat on the relevant date, and

(b) to the extent that it is not included within that value, the biodiversity value of the habitat enhancement which is, on that date, recorded in the register as habitat enhancement to be achieved on the land.

Post-development biodiversity value

8 (1) In relation to any development for which planning permission is granted, the post-development biodiversity value of the onsite habitat is the projected value of the onsite habitat as at the time the development is completed.

(2) That value is to be calculated by taking the pre-development biodiversity value and—

(a) if at the time the development is completed the development will, taken as a whole, have increased the biodiversity value of the onsite habitat, adding the amount of that increase, or

(b) if at the time the development is completed the development will, taken as a whole, have decreased the biodiversity value of the onsite habitat, subtracting the amount of that decrease.

This is subject to paragraph 9.

9 (1) This paragraph applies in relation to any development for which planning permission is granted where—

(a) the person submitting the biodiversity gain plan for approval proposes to carry out works in the course of the development that increase the biodiversity value of the onsite habitat, and

(b) the planning authority considers that the increase is significant in relation to the pre-development biodiversity value.

(2) The increase in biodiversity value referred to in sub-paragraph (1) is to be taken into account in calculating the post-development biodiversity value of the onsite habitat only if the planning authority is satisfied that the condition in sub-paragraph (3) is met.

(3) The condition is that any habitat enhancement resulting from the works referred to in sub-paragraph (1)(a) will, by virtue of—

(a) a condition subject to which the planning permission is granted,

(b) a planning obligation, or

(c) a conservation covenant,

be maintained for at least 30 years after the development is completed.

Registered offsite biodiversity gains

10 (1) "Registered offsite biodiversity gain" means any habitat enhancement, where—

(a) the enhancement is required to be carried out under a conservation covenant or planning obligation, and

(b) the enhancement is recorded in the biodiversity gain site register (as to which, see section 93 of the Environment Act 2020).

(2) References to the allocation of registered offsite biodiversity gain are to its allocation in accordance with the terms of the conservation covenant or planning obligation referred to in sub-paragraph (1)(a).

(3) The biodiversity value of registered offsite biodiversity gain is measured, under the biodiversity metric, in relation to development to which it is allocated.

Biodiversity credits

11 "Biodiversity credits" means credits under section 94 of the Environment Act 2020.

General

12 (1) In relation to development for which planning permission is granted—

"onsite habitat" means habitat on the land to which the planning permission relates;

"planning authority" means the local planning authority, except that—

(a) in a case where the planning permission is granted by Mayoral development order under section 61DB, "planning authority" means such of the Mayor of London or the local planning authority as may be specified in the order;

(b) in a case where the planning permission is granted by the Secretary of State under section 62A, 76A or 77, "planning authority" means such of the Secretary of State or the local planning authority as the Secretary of State may determine;

(c) in a case where the planning permission is granted on an appeal under section 78, "planning authority" means such of the person determining the appeal or the local planning authority as that person may direct.

(2) "Habitat enhancement" means enhancement of the biodiversity of habitat.

(3) References to the grant of planning permission include the deemed grant of planning permission.

PART 2

CONDITION OF PLANNING PERMISSION RELATING TO BIODIVERSITY GAIN

General condition of planning permission

13 (1) Every planning permission granted for the development of land in England shall be deemed to have been granted subject to the condition in sub-paragraph (2).

(2) The condition is that the development may not be begun unless—

(a) a biodiversity gain plan has been submitted to the planning authority (see paragraph 14), and
(b) the planning authority has approved the plan (see paragraph 15).

Biodiversity gain plan

14 (1) For the purposes of paragraph 13(2)(a), a biodiversity gain plan is a plan which—
(a) relates to development for which planning permission is granted, and
(b) specifies the matters referred to in sub-paragraph (2).

(2) The matters are—
(a) information about the steps taken or to be taken to minimise the adverse effect of the development on the biodiversity of the onsite habitat and any other habitat,
(b) the pre-development biodiversity value of the onsite habitat,
(c) the post-development biodiversity value of the onsite habitat,
(d) any registered offsite biodiversity gain allocated to the development and the biodiversity value of that gain in relation to the development,
(e) any biodiversity credits purchased for the development, and
(f) such other matters as the Secretary of State may by regulations specify.

(3) The Secretary of State may by regulations make provision about—
(a) any other matters to be included in a biodiversity gain plan;
(b) the form of a biodiversity gain plan;
(c) the procedure to be followed in relation to the submission of a biodiversity gain plan (including the time by which a plan must be submitted);
(d) persons who may or must submit a biodiversity gain plan.

Approval of biodiversity gain plan

15 (1) For the purposes of paragraph 13(2)(b) a planning authority to which a biodiversity gain plan is submitted must approve the plan if, and only if, it is satisfied as to the matters specified in sub-paragraph (2).

(2) The matters are—
(a) that the pre-development biodiversity value of the onsite habitat is as specified in the plan,
(b) that the post-development biodiversity value of the onsite habitat is at least the value specified in the plan,
(c) that, in a case where any registered offsite biodiversity gain is specified in the plan as allocated to the development—

(i) the registered offsite biodiversity gain is so allocated (and, if the allocation is conditional, that any conditions attaching to the allocation have been met or will be met by the time the development begins), and

(ii) the registered offsite biodiversity gain has the biodiversity value specified in the plan in relation to the development,

(d) that any biodiversity credits specified in the plan as purchased for the development have been so purchased,

(e) that the biodiversity gain objective is met, and

(f) any other matters specified in the plan under paragraph 14(2)(f).

Regulations about determinations

16 The Secretary of State may make regulations as to—

(a) the procedure which a planning authority is to follow in determining whether to approve a biodiversity gain plan (including the time by which a determination must be made);

(b) factors which may or must be taken into account in making such a determination;

(c) appeals relating to such a determination.

Exceptions

17 Paragraph 13 does not apply in relation to—

(a) development for which planning permission is granted—

(i) by a development order, or

(ii) under section 293A (urgent Crown development), or

(b) development of such other description as the Secretary of State may by regulations specify.

Modifications for irreplaceable habitat

18 (1) The Secretary of State may by regulations make provision modifying or excluding the application of this Part of this Schedule in relation to any development for which planning permission is granted where the onsite habitat is "irreplaceable habitat" as defined in the regulations.

(2) Regulations under this paragraph must make provision requiring, in relation to any such development, the making of arrangements for the purpose of minimising the adverse effect of the development on the biodiversity of the onsite habitat.

(3) Regulations under this paragraph may confer powers and duties, including powers and duties in relation to the giving of guidance, on Natural England.

Modifications for particular kinds of planning permission

19 (1) The Secretary of State may by regulations make provision modifying the application of this Part of this Schedule in relation to—

(a) the grant of outline planning permission, where the reservation of matters for subsequent approval has the effect of requiring or permitting development to proceed in phases, or

(b) the grant of any kind of planning permission, where the grant is subject to conditions (whether requiring the subsequent approval of any matters or otherwise) having that effect.

(2) Regulations under this paragraph may include provision for a grant of planning permission referred to in sub-paragraph (1)(a) or (b) to be subject to conditions relating to meeting the biodiversity gain objective referred to in paragraph 2.

20 (1) The Secretary of State may by regulations make provision modifying or excluding the application of this Part of this Schedule in relation to development for which—

(a) planning permission is granted under section 73A (planning permission for development already carried out), or

(b) planning permission is granted by an order under section 102 (orders requiring discontinuance of use etc).

(2) Regulations under this paragraph may in particular include provision—

(a) for paragraph 13 not to apply in relation to the grant of planning permission referred to in sub-paragraph (1)(a) or (b);

(b) for the grant of any such planning permission to be subject to other conditions relating to meeting the biodiversity gain objective.

(3) The conditions referred to in sub-paragraph (2)(b) may include conditions requiring—

(a) habitat enhancement on the land to which the planning permission relates;

(b) the allocation of registered offsite biodiversity gain to any development for which the planning permission is granted;

(c) the purchase of biodiversity credits for any such development.

Further application of this Part

21 The Secretary of State may by regulations make provision to apply this Part of this Schedule in relation to development for which planning permission is granted under section 141 or 177(1), with such modifications or exclusions as may be specified in the regulations."

PART 2

CONSEQUENTIAL AMENDMENTS

3 (1) The Town and Country Planning Act 1990 is amended as follows.

(2) In section 56 (time when development begins), in subsection (3), at the end insert "and paragraph 13 of Schedule 7A".

(3) In section 69 (register of applications etc)—

(a) in subsection (1), at the end insert—

"(e) applications for approval of biodiversity gain plans under Part 2 of Schedule 7A.";

(b) in subsection (2)(a), for "and (aza)" substitute ", (aza) and (e)".

(4) In section 70 (determination of applications: general considerations), in subsection (1)(a), after "section 62D(5)" insert ", paragraph 13 of Schedule 7A".

(5) In section 73 (determination of applications to develop land after non-compliance), after subsection (2A) insert—

"(2B) Nothing in this section authorises the disapplication of the condition under paragraph 13 of Schedule 7A (biodiversity gain condition).

(2C) Subsection (2D) applies where—

(a) for the purposes of paragraph 13 of Schedule 7A a biodiversity gain plan was approved in relation to the previous planning permission ("the earlier biodiversity gain plan"),

(b) planning permission is granted under this section, and

(c) the conditions subject to which the planning permission is granted under this section do not affect the post-development biodiversity value of the onsite habitat as specified in the earlier biodiversity gain plan.

(2D) Where this subsection applies, the earlier biodiversity gain plan is regarded as approved for the purposes of paragraph 13 of Schedule 7A in relation to the planning permission granted under this section."

(6) In section 74A (deemed discharge of planning permission conditions), after subsection (2) insert—

"(2A) But this section does not apply to the condition under paragraph 13 of Schedule 7A (biodiversity gain condition)."

(7) In section 76C (provisions applying to applications made under section 62A), in subsection (2), after "Schedule 1" insert ", or by regulations under paragraph 14(3) or 16 of Schedule 7A,".

(8) In section 84 (simplified planning zone schemes: conditions and limitations on planning permission), at the end insert—

"(5) A simplified planning zone scheme may not disapply the condition under paragraph 13 of Schedule 7A (biodiversity gain condition)."

(9) In section 88 (enterprise zones), after subsection (3) insert—

"(3A) Subsection (3) is subject to paragraph 13 of Schedule 7A (biodiversity gain condition)."

(10) In section 96A (power to make non-material changes to planning permission), after subsection (3) insert—

"(3A) The conditions referred to in subsection (3)(b) do not include the condition under paragraph 13 of Schedule 7A (biodiversity gain condition)."

(11) In section 97 (revocation or modification of planning permission), at the end insert—

"(7) Subsection (1) does not permit the revocation or modification of the condition under paragraph 13 of Schedule 7A (the biodiversity gain condition), subject as follows.

(8) The Secretary of State may by regulations make provision—

(a) for the condition under paragraph 13 of Schedule 7A to apply in relation to the modification of planning permission under this section, subject to such modifications as may be specified in the regulations;

(b) for planning permission modified under this section to be subject to other conditions relating to meeting the biodiversity gain objective referred to in paragraph 2 of Schedule 7A (including conditions of a kind referred to in paragraph 20(3) of that Schedule)."

(12) In section 100ZA (restriction on power to impose planning conditions in England), in subsection (13)(c), after "limitation" insert "but do not include the condition under paragraph 13 of Schedule 7A (biodiversity gain condition)".

(13) In section 106 (planning obligations), in subsection (1), in the words before paragraph (a), after "106C" insert ", Schedule 7A".

(14) In section 106A (modification and discharge of planning obligations), after subsection (6) insert—

"(6A) Except in such cases as may be prescribed, the authority may not under subsection (6) discharge or modify the planning obligation if the authority considers that doing so would—

(a) prevent the biodiversity gain objective referred to in paragraph 2 of Schedule 7A from being met in relation to any development, or

(b) give rise to a significant risk of that objective not being met in relation to any development."

(15) In section 333 (regulations and orders), after subsection (3A) insert—

"(3AA) No regulations may be made under paragraph 2(4) of Schedule 7A (biodiversity gain condition) unless a draft of the instrument containing the regulations has been laid before, and approved by a resolution of, each House of Parliament."

ENVIRONMENT BILL

EXPLANATORY NOTES

What these notes do

These Explanatory Notes relate to the Environment Bill as introduced in the House of Commons on 30 January 2020 (Bill 9).

- These Explanatory Notes have been prepared by the Department for Environment, Food and Rural Affairs (Defra) in order to assist the reader of the Bill and to help inform debate on it. They do not form part of the Bill and have not been endorsed by Parliament.
- These Explanatory Notes explain what each part of the Bill will mean in practice; provide background information on the development of policy; and provide additional information on how the Bill will affect existing legislation in this area.
- These Explanatory Notes might best be read alongside the Bill. They are not, and are not intended to be, a comprehensive description of the Bill.

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Overview of the Bill

1 The Environment Bill ("the Bill") comprises two thematic halves. The first provides a legal framework for environmental governance. The second makes provision for specific improvement of the environment, including measures on waste and resource efficiency, air quality and environmental recall, water, nature and biodiversity, and conservation covenants.

2 The first part of the Bill was published in part as the draft Environment (Principles and Governance) Bill on 19 December 2018, fulfilling a legal obligation set out in section 16 of the European Union (Withdrawal) Act 2018. The measures published at that time related only to environmental principles and governance, and placing the previous government's 25 Year Environment Plan on a statutory footing.

3 The remaining parts of the Bill make provision for a range of environmental improvements. In its 2019 Manifesto, *Get Brexit Done: Unleash Britain's Potential*, the government pledged to "protect and restore our natural environment after leaving the EU". Measures in the Bill – many of which were consulted on by the previous government and included in the Environment Bill introduced late in the last parliament – take legislative steps to deliver that commitment. This supports the ambition to have 'the most ambitious environmental programme of any country on earth'.

4 The Environmental Governance Part of the Environment Bill (Part 1) includes provisions to:

- allow government to set long-term targets (of at least 15 years duration) in relation to the natural environment and people's enjoyment of the natural environment via statutory instrument;
- require government to meet long-term targets, and to prepare remedial plans where long-term targets are not met;
- require government to set, by October 2022, at least one long-term target in each of the priority areas of air quality, water, biodiversity, and resource efficiency and waste reduction;
- require government to set and meet an air quality target for fine particulate matter in ambient air ($PM_{2.5}$);
- require government to periodically review all environmental targets to assess whether meeting them would significantly improve the natural environment in England;
- establish the process by which a long-term target is set and amended, as well as an enhanced process where a long-term target is lowered or revoked;
- require the government to have, and maintain, an Environmental Improvement Plan, a plan to significantly improve the natural environment, which sets out the steps the government intends to take to improve the natural environment, and which sets out interim targets towards meeting the long-term targets;
- require government to produce an annual report on the Environmental Improvement Plan, to consider progress towards improving the natural environment and meeting the targets;

- require government to review the plan periodically, to consider progress and whether further or different steps are needed to improve the natural environment and meet the targets, and if appropriate revise the plan;
- require government to collect and publish data used to measure progress in improving the natural environment and meeting the targets;
- require the publication of a policy statement on environmental principles setting out how environmental principles specified under the Bill are to be interpreted and applied by Ministers of the Crown during the policymaking process;
- create a new, statutory and independent environmental body, the Office for Environmental Protection (OEP), to hold government to account on environmental law and its Environmental Improvement Plan once the UK leaves the EU;
- define the scrutiny, complaints and enforcement functions of the OEP and their scope;
- establish an OEP enforcement process of environmental review in the Upper Tribunal;
- define the nature of the OEP, including considerations of membership, remuneration, staffing, powers, reporting, funding, accounts and other issues;
- require the government to publish a report on the impact of all new environmental primary legislation; and
- require the government to undertake a report on environmental legislation across the world on a two yearly basis.

5 The Environmental Governance: Northern Ireland Part of the Environment Bill (Part 2) includes provisions to:

- extend the application of the OEP to Northern Ireland, and make separate provision for Environmental Improvement Plans and environmental principles in Northern Ireland.

6 The Waste and Resource Efficiency Part of the Environment Bill (Part 3) includes provisions to:

- require producers to pay the full net cost of managing their products at end of life to incentivise more sustainable use of resources;
- allow deposit return schemes to be established, whereby a deposit is included in the price of an in-scope item (such as a drink in a bottle or can) which is redeemed when the item is returned to a designated point;
- enable producer responsibility obligations to be applied at all levels of the waste hierarchy to, for example, facilitate the prevention of food waste and increase the redistribution of food surplus;
- enable charges to be applied to specified single-use plastic items;

- require local authorities in England to collect the same range of materials for recycling from households;
- ensure households have a weekly separate food waste collection;
- ensure businesses and public bodies present recyclable materials for separate collection and arrange for its separate collection;
- enable government to set resource efficient product standards and information and labelling requirements, to drive a shift in the market towards durable, repairable and recyclable products;
- improve the proportionality and fairness of litter enforcement, by issuing statutory guidance on the use of enforcement powers and extending an existing power to set out conditions to be met by all those carrying out enforcement activity;
- improve the management of waste, by enabling the Secretary of State to make regulations in relation to waste tracking digitally;
- improve the regulators' effectiveness in tackling waste crime, reducing the cost of that criminal activity on the wider economy, environment and society;
- allow the Environment Agency to be more flexible and responsive in managing exempt waste sites and ensure proportionate controls are in place to avoid environmental harm or illegal activity as waste market practices change;
- fill a gap in existing powers to ensure that waste can be collected and disposed of when normal processes fail;
- enable the Secretary of State to make regulations to amend the permitted range of penalties for existing Fixed Penalty Notices; and
- enable the Secretary of State to regulate the import, export or transit of waste and hazardous waste.

7 The Air Quality and Environmental Recall Part of the Environment Bill (Part 4) includes provisions to:

- amend Part 4 of the Environment Act 1995 (which creates the Local Air Quality Management Framework) to strengthen the requirements in respect of the National Air Quality Strategy, including a requirement for it to be regularly reviewed;
- amend the Local Air Quality Management Framework to clarify duties and enable greater cooperation between different levels of local government, and other relevant public bodies, in the preparation of Local Air Quality Action Plans;
- amend Part 3 of the Clean Air Act 1993 to enable quicker, simpler and more proportionate enforcement of Smoke Control Areas, a key means by which local authorities can control pollution from domestic solid fuel burning; and
- provide for mandatory recall notices for vehicles and equipment that do not

comply with relevant environmental standards and for fines to be issued when a minimum recall rate is not met.

8 The Water Part of the Environment Bill (Part 5) includes provisions to:

- change the procedural requirements for Water Resources Management Plans, and enable increased collaboration between different water undertakers to better manage water resources;
- require the preparation of Drainage and Sewerage Management Plans by sewerage undertakers, to better plan for the management of waste water;
- modernise the process for modification of water and sewerage undertaker licence conditions by the Water Services Regulation Authority ("Ofwat") to bring it in line with other utilities, and to strengthen Ofwat's ability to improve water and sewerage undertakers' operations;
- change the circumstances in which a licence to abstract water from the environment can be revoked or varied without paying compensation, to prevent damage to the environment;
- enable future updates to the lists of priority substances in water quality legislation, and enable the reallocation of regulatory responsibilities in the Solway Tweed river basin district; and
- enable updates to be made to the valuation calculations relevant to the apportionment of internal drainage board (IDB) charges in secondary legislation, allowing for the creation of new or expansion of existing IDBs where there is a local desire to do so.

9 The Nature and Biodiversity Part of the Environment Bill (Part 6) includes provisions to:

- amend section 40 of the Natural Environment and Rural Communities Act 2006 to strengthen and improve the duty on public bodies to conserve and enhance biodiversity, in accordance with the proper exercise of their functions;
- mandate net gain in biodiversity through the planning system, requiring a 10% increase in biodiversity after development, compared to the level of biodiversity prior to the development taking place, as measured by a metric set out by Defra;
- require the preparation and publication of Local Nature Recovery Strategies, a tool to direct action for nature, and place an emphasis on supporting local leadership of nature improvement; and
- provide greater enforcement powers to the Forestry Commission to reduce illegal tree felling, and require local authorities to consult local residents prior to the felling of street trees.

10 The Conservation Covenants Part of the Environment Bill (Part 7) includes provisions to:

- provide for Conservation Covenants: voluntary, legally binding private agreements between landowners and responsible bodies, designated by the

Secretary of State, which conserve the natural or heritage features of the land, enabling long-term conservation.

11 The Miscellaneous and General Provisions Part of the Environment Bill (Part 8) includes provision to:

- amend two pieces of retained European Union law relating to the regulation of chemicals.
- allow for consequential provision; regulations; commencement and transitional or saving provision; and
- set out the position in relation to Crown application; financial provisions; and the extent and the short title of the Bill, which may be cited as the Environment Act 2019.

Policy background

Exiting the European Union (EU)

12 On 1 January 1973, the UK joined the European Economic Community, now the European Union.

13 On 17 December 2015, the European Union Referendum Act 2015 received Royal Assent. The Act made provision for holding a referendum in the UK and Gibraltar on whether the UK should remain a member of the EU. The referendum was held on 23 June 2016 and a majority voted to leave the EU.

14 The European Union (Notification of Withdrawal) Act 2017 received Royal Assent on 16 March 2017. On 29 March 2017, the Prime Minister gave notification of withdrawal of the UK from the EU under Article 50(2) of the Treaty on European Union. The European Union (Withdrawal) Act 2018 received Royal Assent on 26 June 2018. Section 16 of the Withdrawal Act required the Secretary of State to publish a draft Bill to make provision for a new environmental governance body and a requirement for Ministers of the Crown to have regard to a new policy statement on environmental principles when making policy following the UK's withdrawal from the EU.

15 The European Union (Withdrawal Agreement) Act 2020 received Royal Assent on 23 January 2020, ratifying the Brexit Withdrawal Agreement.

Part 1: Environmental Governance

16 The Draft Environment (Principles and Governance) Bill 2018 was published for parliamentary pre-legislative scrutiny on 19 December 2018, fulfilling requirements for publication of a draft Bill under section 16 of the European Union (Withdrawal) Act 2018. Part 1 of this Bill updates that Draft Bill in light of pre-legislative scrutiny reports by the Environment, Food and Rural Affairs Select Committee and Environmental Audit Committee in the previous parliament, which were published on 30 April 2019 and 24 April 2019 respectively. The government of the day responded to these reports when it introduced its Environment Bill, which the current Bill largely takes forward, in October 2019.

17 Most of the UK's environmental law and policy derives from the EU, and EU structures and

- modernising water regulation by reforming elements of the abstraction licensing regime to link it more tightly to the government's objectives for the water environment; and

- enabling updates to be made to the valuation calculations relevant to the apportionment of internal drainage board (IDB) charges in secondary legislation, allowing for the creation of new or expansion of existing IDBs where there is a local desire to do so.

43 It also includes measures to protect water quality in our surface and groundwater, by enabling updates to the lists of priority substances that pose a threat to water bodies in line with the latest scientific knowledge, in the absence of powers under section 2(2) of the European Communities Act 1972.

Part 6: Nature and Biodiversity

44 Nature is in decline, much of England's wildlife is deteriorating, and many ecosystems are degraded.[1] The UK has a number of international and legislative commitments to take urgent and effective action to halt the loss of nature or biodiversity.

45 Since the 25 Year Environment Plan set the ambition towards embedding a broad 'environmental net gain' principle in the planning system, government has focussed on embedding the principle of biodiversity net gain. In July 2018, the revised National Planning Policy Framework strengthened planning policy on biodiversity net gain by making it clearer that all development in scope should deliver biodiversity net gains. From December 2018 to February 2019, government consulted on whether biodiversity net gain should be made mandatory and received 470 responses. In March 2019, in response to strong support at consultation, the previous government committed to mandating biodiversity net gain in the Environment Bill. The government committed in its manifesto to increasing biodiversity whilst continuing progress towards a target of 300,000 new homes a year by the mid-2020s.

46 Section 40 of the Natural Environment and Rural Communities (NERC) Act 2006 requires all public authorities carrying out functions in England (together with HMRC in Wales also) to have regard to conserving biodiversity when delivering their functions. The existing wording does not adequately reflect the ambition or language of the 25 Year Environment Plan. Shifting the focus of the duty to an active requirement to seek the further conservation and enhancement of nature should better align public authorities' action on biodiversity with government's ambition.

47 Spatial plans enable the public, private and charity sectors to direct investment in nature to where it can best benefit the natural environment, and have an important role to play in delivering the government's commitment to nature recovery. Although such plans do exist in some areas of England, they are often produced by a variety of bodies working at different spatial scales. Local Nature Recovery Strategies (LNRSs) will put spatial planning for nature on a statutory footing, and will support local action by consistently mapping

[1] JNCC. 2019. Sixth National Report to the United Nations Convention on Biological Diversity: United Kingdom of Great Britain and Northern Ireland. JNCC, Peterborough. Available at http://archive.jncc.gov.uk/pdf/UK_CBD_6NR_v2.pdf.

important existing habitats and opportunities to create or restore habitat. For example, the biodiversity net gain consultation identified a need for local plans for nature to target biodiversity increases. Developed through a collaborative approach, LNRSs will also support the delivery of a Nature Recovery Network by acting as a key tool to help local partners better direct investment and action that improves, creates and conserves wildlife-rich habitat.

48 This Bill also includes measures covering forestry and street trees, including amendments to the Forestry Act 1967 to tackle illegal felling, and measures requiring local highway authorities to consult the public before felling any street trees. Current regulations on tree felling include felling licences in the Forestry Act 1967, provisions in the Highways Act 1980, Tree Preservation Orders (TPOs) under the Town and Country Planning Act 1990, and the Equalities Act 2010. On 30 December 2018, the previous government announced a public consultation on protecting and enhancing England's trees and woodlands which received 4671 responses.

Part 7: Conservation Covenants

49 This Part addresses the absence of a simple legal tool that landowners can use to secure conservation benefits when land is sold or passed on. Conservation covenants are private, voluntary agreements between a landowner and responsible body, such as a conservation charity or public body. They provide for the conservation of the natural environment and heritage assets for the public good. They can bind subsequent owners of the land, so have the potential to deliver long-lasting conservation benefits.

50 In the 25 Year Environment Plan, government set out its ambition of recovering nature. Individual landowners can play an important role in conservation efforts, but under the current law it is difficult to ensure that legal obligations for conservation survive once the land has been sold or passed on. As a result, conservation opportunities are missed or fail to secure long-term, sustainable outcomes. Complex, legal workarounds have sometimes been used but these can be costly and do not always appeal to landowners. Conservation covenants are used in other countries including New Zealand, the USA, Canada and Scotland.

51 The Law Commission examined the need for conservation covenants and concluded that legislation should be introduced, preparing a draft Bill in 2014. The previous government consulted on the Law Commission proposals in February and March 2019. There was broad support for conservation covenants from the 112 responses across a range of stakeholders. The previous government committed to include provisions for conservation covenants in the Environment Bill in its consultation response published on 23 July 2019. The key change to the Law Commission Bill is to allow for-profit bodies to apply to become responsible bodies.

Part 8: Miscellaneous and General Provisions

52 The use of chemicals in the EU is regulated by Regulation (EC) No 1907/2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), establishing a European Chemicals Agency (the "REACH Regulation"). The REACH Regulation will form part of retained EU law by virtue of the European Union (Withdrawal) Act 2018 ("the Withdrawal Act"). The REACH Enforcement Regulations 2008 provide for the contravention of various provisions of the REACH Regulation to be a criminal offence, and set out which domestic bodies enforce those offences. Part 8 and Schedule 19 give the Secretary of State the power to amend the REACH Regulation and the REACH Enforcement Regulations 2008 as they apply in the UK after exit day, in order to keep them up to date

through actions for species and habitats.

839 Subsection (7) provides that in the case of HMRC, the duty does not apply to its functions in England alone, but also extends to its functions in Wales. This is to maintain the amendment made to section 40 by the Environment (Wales) Act 2016. This subsection also clarifies the spatial extent of the general biodiversity duty, establishing that the territorial sea adjacent to England falls under the duty.

840 Subsection (8) amends section 41 of the NERC Act so that its references to section 40 correctly reflect the amendments made by this Bill.

Clause 94: Biodiversity reports

841 This clause adds new section 40A to the NERC Act, which creates a power for the Secretary of State to designate public authorities who are required to report on the action they have taken under the biodiversity objective set out in clause 93. It also defines, at a high level, the content of the biodiversity reports and their frequency. These reports will capture how public authorities with significant landholdings have sought to conserve and enhance biodiversity, and will contribute to the improvement of information on protected sites, priority habitats and priority species.

New section 40A Biodiversity reports

842 Subsection (1) sets out the public authorities to which new section 40A applies. The scope of this reporting requirement is limited in comparison to the general biodiversity duty, as the latter applies to all public authorities. All local authorities and local planning authorities, excluding parish councils, will be required to produce biodiversity reports. It would not be reasonable to require other public authorities with few or no landholdings to produce reports, and so additional public bodies that must report will be designated by the Secretary of State in subsequent regulations as stipulated in subsection (8)(a).

843 Subsection (2) requires the public authorities identified in subsection (1) to produce biodiversity reports.

844 Subsections (3) and (4) specify the required content of a biodiversity report. In subsection (3), paragraphs (a) and (b) set out that this report will both reflect on the action the public authority has taken under the duty over the reporting period, and look forward to the actions it will take in the subsequent five-year reporting period. This qualitative information will be accompanied by quantitative data, as provided by paragraph (c). The specific data required will be stipulated in regulations issued by the Secretary of State; it could, for example, include the proportion of local wildlife sites in favourable condition. In addition to the required qualitative and quantitative information, paragraph (d) sets out that the public authority is free to include any other information it considers relevant. Subsection (4) specifies additional reporting requirements for local planning authorities – namely, the provision of information on biodiversity net gain. Local planning authorities must give a qualitative summary of the action they have taken to comply with mandatory biodiversity net gain, and any gains within their jurisdiction.

845 Subsections (5) to (7) set out the timing and frequency of reporting. Under subsection (5), the report must be published within 12 weeks after the end of the period that it covers, in order to ensure that the information shared is up-to-date and relevant upon publication. Subsection (6) sets out that a public authority must publish its first biodiversity report within three years of being designated as required to report, although it is at liberty to decide when to publish within this window. This applies both in cases when the public authority is designated on the introduction of clause 94 and when the public authority is brought into scope at a later point. After the first report, subsection (7) establishes that the

public authority must publish subsequent reports at least every five years, though again it can choose when to publish within this window.

846 Subsection (8) creates a power for the Secretary of State to issue regulations that designate public authorities as required to report, and to further define what data must be included in the biodiversity report. This will ensure key quantitative data is reported in a consistent fashion across all reports, thereby making comparisons across the reports easier. Having such data defined in regulations will also allow for it to be updated in the future as required.

847 Subsection (9) clarifies that the reporting requirement will not apply to public authorities that do not exercise functions in England.

848 Subsections (10) and (11) establish that the regulations should be issued through a statutory instrument subject to the negative resolution procedure in Parliament.

849 Subsection (12) clarifies that the definitions within this section are the same as used in the wider section 40 duty and in clause 93.

Clause 95: Local nature recovery strategies for England

850 This clause provides for the creation of Local Nature Recovery Strategies (LNRSs) in England, for how the geographical coverage of each LNRS will be determined, and for the relationship between LNRSs and the biodiversity duty under the NERC Act.

851 Subsections (1) and (2) set out that the requirement for LNRSs applies to England and that, taken together, all areas of England will be covered by an LNRS.

852 Subsection (3) sets out that the area covered by each LNRS will be determined by the Secretary of State. This approach allows flexibility to make sure that the area covered by each LNRS balances the potentially competing needs to be ecologically meaningful (that is, based on the common geographic, geological, topographical or other physical features that help define the ecology of an area, and of sufficient scale to include a range of habitats and land uses) and align with existing administrative boundaries. It will also allow alignment of boundaries between neighbouring LNRS areas to avoid gaps or overlap. It is anticipated that the area covered by each LNRS will be approximately county-scale, but with the potential to vary to best accommodate differing local circumstances.

853 Establishment of the area covered by an LNRS is closely linked to agreement on the identity of the responsible authority, which will lead on the production of the LNRS. The process for doing this is set out in clause 96.

854 Subsection (4) states that the area of a local authority, other than a county council, may not be split between different LNRSs. This requirement reflects the importance of local authorities in producing and implementing the LNRS, and is intended to make it easier for individual local authorities to do so by making each LNRS area an aggregate of local authority areas. This requirement will also facilitate the agreement of individual LNRS boundaries, and make sure they operate at scales most relevant to local authorities.

855 The bodies included within the definition of "local authority" are listed in clause 96(2). County councils are exempted from the general prohibition in splitting local authorities between LNRSs because of their larger size, and the potential that not doing so would result in a *de facto* requirement that LNRSs conform to county boundaries, irrespective of ecological or other practical considerations. Whilst in many cases there may be good reason for LNRS and county council boundaries to align, it is reasonable to anticipate situations where this would not be the case.

856 Subsection (5) highlights the relationship between LNRSs and the duty to conserve

appeal procedure.

1546 Sub-paragraph (1) sets out how the CMA's costs for dealing with the appeal will be payable.

1547 Sub-paragraph (2) sets out who is liable to pay the CMA's costs, depending on the outcome of the appeal.

1548 Under sub-paragraph (3), the CMA can also order one party to pay the reasonable costs of the other party, depending on the appeal outcome.

1549 Under sub-paragraph (4), any party required to pay costs to another party must do so within 28 days from the day after the date of the CMA order.

1550 Under sub-paragraph (5), where the costs are not paid to the other party within the time period, they will accrue interest as set out in the order.

1551 Under sub-paragraph (6), any unpaid costs may be recovered as a civil debt by the person entitled to the costs.

1552 Paragraph 13 of new Schedule 2ZA sets out the definitions of terms used.

Schedule 14: Biodiversity gain as condition of planning permission

Part 1: Biodiversity gain condition

1553 Paragraph 1 inserts new section 90A into the Town and Country Planning Act 1990, which gives effect to new Schedule 7A.

1554 Paragraph 2 inserts new Schedule 7A.

New Schedule 7A Biodiversity gain in England

Part 1: Overview and interpretation

1555 Paragraph 1 of Schedule 7A gives effect to paragraphs 2 to 11, which set out details of a new general condition to all planning permissions granted in England, subject to exceptions. The condition requires a biodiversity gain plan to be submitted and approved by the planning authority before development can lawfully commence. The biodiversity gain plan should contain an assessment of the value of natural habitats before development and after development, and ensure that at least a 10% net gain is achieved between the earlier and later values. The Town and Country Planning Act 1990 already allows for planning permission to be granted subject to condition(s). This "general condition" for biodiversity gain, which is mandatory for all planning permissions, is novel. Because the condition is deemed to have been granted, it exists in statute prior to the grant of planning permission. It may therefore be met at the time of granting planning permission where the planning authority also approves a biodiversity gain plan, meaning that the biodiversity net gain plan may not need to be submitted and approved in a separate process after planning permission has been granted. This could be used for straightforward planning applications where the relevant information is available upfront. The general condition will not apply to all development in all scenarios. Part 1 includes a power to detail these exceptions in secondary legislation.

1556 Paragraph 2 sets out the biodiversity gain objective and how it must be met. The objective is that the biodiversity value, expressed in biodiversity units, attributable to the development exceeds that which existed before development by at least 10%. Sub-

paragraph (4) gives the Secretary of State a power to vary the percentage gain required.

1557 Paragraph 3 identifies the biodiversity metric as the approach which will be used to calculate the relative biodiversity value of any habitat. The metric uses habitats as a proxy for biodiversity value, which it measures and expresses in terms of 'biodiversity units'.

1558 Paragraph 4 makes provision for the Secretary of State to publish the biodiversity metric, the tool which is used to measure the relative biodiversity value of habitats as relevant to this schedule. Sub-paragraphs (3) and (5) give the Secretary of State the power to update the biodiversity metric, and set out any arrangements for transition when the metric is updated so that developers and planning authorities are clear what is required where, for example, a planning application is under consideration on the date the updated version of the metric comes into effect. Updates to the metric will allow technical improvements, reflecting improved ecological understanding and further evaluation of the metric's application in practice, to the metric to be incorporated into the approach. Updates will be infrequent to avoid creating unnecessary uncertainty for the planning system. The intention is to publish a timeline of planned updates. The provision also enables the Secretary of State to make transitional provision where the metric is revised and republished.

1559 Paragraph 5 defines the date on which the pre-development biodiversity value of land should be taken to be measured. Sub-paragraph (3) enables a developer and a local planning authority to agree another date where more appropriate than the default set out in sub-paragraph (2). Sub-paragraph (4) makes provision for circumstances in which the planning authority is aware that there has been degradation of habitats in advance of development.

1560 Paragraph 6 sets out that the pre-development biodiversity value of on-site habitats is to be taken as the value before certain activities took place. These activities must have been carried out without planning permission, or without an alternative permission specified by the Secretary of State in regulations, on or after 15 October 2019. The activities must also have reduced the biodiversity value of habitats on the site below what it would otherwise have been at the time of planning application or planning permission.

1561 Paragraph 7 defines the pre-development biodiversity value in the event that a site registered as a "biodiversity gain site" is developed. It sets out that the pre-development value of the land should be taken to be the enhanced value of the registered site, regardless of whether or not the registered biodiversity enhancement has in fact been delivered successfully. This will mean that the development of a registered compensation site, whilst unlikely and undesirable, should not undermine the overall biodiversity gain outcome.

1562 Paragraph 8 defines the post-development biodiversity value of habitat on the development site as the projected value of habitats on the development site. The value needs to be projected because a planning authority will need to use this figure before development starts to determine whether the development will achieve the net gain objective. In practice, the post-development biodiversity value of habitats on the development site will be determined by applying the metric to the developer's plan for the development site as detailed in the biodiversity gain plan.

1563 Paragraph 9 stipulates that significant increases in onsite biodiversity value can only be considered part of the post-development biodiversity value if they are secured through a suitable mechanism and will be maintained for at least 30 years after the completion of development.

1564 Paragraph 10 defines what can be counted towards "registered offsite biodiversity gain" in relation to a development. This relates to gain achieved on land other than the development site. Where a developer makes an agreement with a third party to do so, or

enters into an agreement to do so themselves, this gain can be allocated to the development to be counted towards meeting the biodiversity objective. The biodiversity gain and its allocation to a development must be recorded on the biodiversity gains site register.

1565 Paragraph 11 makes reference to biodiversity credits, as described in clause 90 of the current Bill.

1566 Paragraph 12 defines the terms "developer", "onsite habitat" and "planning authority" as they are applied in the new Schedule 7A. Sub-paragraph (2) states that references to planning permission should be taken to include deemed planning permission, which is granted by government departments to certain development. This includes development approved under the Electricity Act 1989 and the Transport and Works Act 1989.

Part 2: Condition of planning permission relating to biodiversity gain

1567 Paragraph 13 sets out and applies the general condition to all planning permissions granted for development in England, subject to exceptions in paragraphs 17 and 18 and further application beyond the standard definition of planning permission in paragraph 19.

1568 Paragraph 13 contains the wording of the general condition itself, which requires that a developer wishing to commence a development approved by planning permission will first need to submit a biodiversity gain plan to the planning authority, as defined in paragraph 12, and obtain the planning authority's approval of this plan.

1569 Paragraph 14 establishes what information a biodiversity gain plan must include to satisfy the general condition. A biodiversity gain plan must specify all the information necessary for a planning authority to be able to approve the plan under paragraph 15. This includes information about the biodiversity value of habitats before and after development and steps taken to minimise harm to habitats during development.

1570 The Secretary of State may specify other matters to be included in the plan under sub-paragraph (2)(f). Sub-paragraph (3) provides for the Secretary of State to specify the procedure which must be followed when submitting a gain plan.

1571 Paragraph 15 requires planning authorities to approve a biodiversity gain plan if, and only if, key information in the biodiversity gain plan is accurate and the biodiversity gain objective is met. Authorities need to be satisfied with the information provided, including the value of onsite and offsite habitat gains and losses associated with the development. If the authority decides they are not satisfied, development would not be able to proceed lawfully unless the developer successfully appeals this decision.

1572 Paragraph 16 gives the Secretary of State the power to specify procedures to follow and factors to be taken into account when approving a biodiversity gain plan, and in relation to appeals against decisions.

1573 Paragraph 17 exempts all development granted planning permission by the Secretary of State using a development order, or under provisions for urgent Crown development, from the application of the general condition. This includes development granted permission by the General Permitted Development Order, which allows various types of development to proceed without requiring a planning application. Paragraph 17 also gives the Secretary of State the power to exempt development from the requirement to apply the general condition.

1574 In recognition of the fact that some habitat is irreplaceable, and therefore impossible to achieve a net gain on, paragraph 18 allows the Secretary of State to make regulations that modify or exclude the application of the general condition to irreplaceable habitat. Sub-

paragraph (2) specifies that, where development does impact irreplaceable habitats, regulations must require measures to be taken to minimise the negative impacts of this development on those habitats, and that those measures should be agreed with the planning authority where they are not made by the planning authority in the first place. Sub-paragraph (3) allows regulations to confer powers and duties on Natural England in relation to giving guidance on the treatment of irreplaceable habitat in development.

1575 Whilst it is generally agreed in practice that development cannot claim biodiversity net gain in cases when development results in land take from statutory protected sites (such as Sites of Special Scientific Interest, Special Protection Areas, Special Areas of Conservation, and Ramsar sites), development on such sites is not specifically exempted from the net gain requirement. The biodiversity metric does not address impacts on species, recognise the significance of site designations, take account of indirect impacts, cumulative impacts or in-combination impacts. In recognition of these limitations, the biodiversity net gain requirement for development on such sites is additional to any existing legal or policy requirements for statutory protected areas and their features, including restoration and conservation of designated features and the achievement of favourable conservation status and favourable condition. These requirements will need to be dealt with separately by the developer and planning authority.

1576 Paragraph 19 allows the Secretary of State to modify the application of this Part to permissions for phased development or developments where subsequent approval has the effect of phasing development. Sub-paragraph (2) sets out that regulations may include provisions for biodiversity gain conditions to be included in the grant of planning permissions for these types of permission.

1577 Paragraph 20 gives the Secretary of State powers to modify or exclude the application of this Part to planning permissions for development already carried out, or planning permissions granted for the alteration or removal of buildings or works by any order requiring discontinuance of any use of land.

1578 Paragraph 21 makes provision for the Secretary of State to make provision as to the application of this Part of the Schedule to planning permission granted in relation to a purchase notice and the successful appeal of a planning enforcement notice.

Part 2: Consequential amendments.

1579 Paragraph 3 details the consequential amendments to the Town and Country Planning Act 1990.

Schedule 15: Controlling the felling of trees in England

Introductory

1580 This Schedule amends Part 2 of the Forestry Act 1967 ("the 1967 Act").

Penalty for felling without licence: increase of fine

1581 Paragraph 2 amends section 17 to preserve in Wales the current level of fine that the Courts are able to impose (level 4 (£2,500) or twice the value of the tree, whichever is the higher) following conviction for a section 17 offence (felling without a licence where one was required), whilst amending the fine available in respect of such a conviction in England to an unlimited level 5 fine.

Restocking notices to be local land charges

1582 Paragraph 3 amends section 17A by inserting new subsection (1B), which provides that a

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"Managing Natura 2000 sites
The provisions of Article 6 of the 'Habitats' Directive 92/43/EEC"

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5.4.4. *Key elements to consider in the compensation measures*

The compensatory measures under Article 6(4) must address all issues, be they technical, legal or financial, needed to offset the negative effects of a plan or project and to maintain the overall coherence of the Natura 2000 network.

The following list provides an overview of elements to consider:

- Tight coordination and cooperation between Natura 2000 authorities, assessment authorities and the proponent of the plan or project.
- Clear objectives and target values according to the site's conservation objectives.
- Description of the compensatory measures, accompanied by a scientifically robust explanation of how they will effectively compensate for the negative effects of the plan or project on the species and habitats affected in light of the site's conservation objectives, and how they will ensure that the overall coherence of Natura 2000 is protected.
- Demonstration of the technical feasibility of the measures in relation to their objectives.
- Demonstration of the legal and/or financial feasibility of the measures according to the timing required.
- Analysis of suitable locations and acquisition of the rights (purchase, lease...) to the land to be used for the compensatory measures.
- Explanation of the time-frame in which the compensation measures are expected to achieve their objectives.
- Timetable for implementation and co-ordination with the schedule for the plan or project implementation.
- Public information and/or consultation stages.
- Specific monitoring and reporting schedules based on progress indicators according to the objectives of compensation measures.
- The financing programme approved during the necessary period to guarantee the success of the measures.

5.5. CRITERIA FOR DESIGNING COMPENSATORY MEASURES

5.5.1. *Targeted compensation*

Compensatory measures under the Habitats Directive must be established according to reference conditions that are defined after the description of the integrity of the site likely to be lost or deteriorated, and according to the likely significant negative effects that would remain after mitigation.

Once the integrity of the site likely to be damaged and the actual extent of the damage have been identified, the compensatory measures must address these issues specifically, so that the elements of integrity contributing to the overall coherence of the Natura 2000 network are compensated for in the long term. Thus, these measures should be the most appropriate to the type of impact predicted and should be focused on objectives and targets clearly addressing the Natura 2000 elements affected. They must clearly refer to the structural and functional aspects of the site integrity, and the related types of habitats and species populations that are affected.

This entails that the compensatory measures must necessarily consist of ecological measures. Therefore, payments to individuals or towards special funds, regardless of whether or not these are ultimately allocated to nature conservation projects are not suitable under the Habitats Directive. In addition, any secondary or indirect measure that might be proposed to enhance the performance of the core compensatory measures must have a clear relationship to the objectives and targets of the compensatory measures themselves.

As an example, in designing compensatory measures for species, there is a need to identify:

- the species adversely affected, their total numbers and the proportion of the total population(s) that these occur in;
- the principal function(s) of the habitats that will be adversely affected that the species depend on e.g. feeding, roosting, etc.;
- the measures needed to compensate for the damage to the habitat functions and species affected so that they are restored to a state that reflects the favourable condition of the area affected.

5.5.2. *Effective compensation*

The feasibility and effectiveness of compensatory measures are critical to the implementation of Article 6(4) of the Habitats Directive, in keeping with the precautionary principle and good practice. In ensuring effectiveness, technical feasibility must go hand in hand with the appropriate extent, timing and location of the compensatory measures.

Compensatory measures must be feasible and operational in reinstating the ecological conditions needed to ensure the overall coherence of the Natura 2000 network. The estimated timescale and any maintenance action required to enhance performance should be known and/or foreseen right from the start before the measures are rolled out. This must be based on the best scientific knowledge available, together with specific investigations for the precise location where the compensatory measures will be implemented. Measures for which there is no reasonable guarantee of success should not be considered under Article 6(4), and the likely success of the compensation scheme should influence the final approval of the plan or project in line with the prevention principle. In addition, when it comes to deciding between different possibilities for compensation, the most effective options, with the greatest chances of success, must be chosen.

The programme of compensatory measures needs to include detailed monitoring during implementation to ensure effectiveness in the long term. Being in the framework of the

Natura 2000 network, such monitoring should be co-ordinated with, and possibly integrated into monitoring under Article 11 of the Habitats Directive.

Measures showing in practice a low level of effectiveness in contributing to the objectives should be modified accordingly.

5.5.3. *Technical feasibility*

According to current knowledge, it is highly unlikely that the ecological structure and function or the related habitats and species populations can be reinstated to the status they had before the damage by a plan or project. To overcome the intrinsic difficulties standing in the way of full success for the ecological conditions, the design of compensatory measures must:

(1) follow scientific criteria and evaluation in accordance with best scientific knowledge, and

(2) take into account the specific requirements of the ecological features to be reinstated (e.g. soil, humidity, exposure, existing threats and other conditions critical to the success of reinstatement).

The aspects critical to technical feasibility will determine the suitability of the location of compensatory measures (spatial feasibility), the appropriate timing and their required extent.

In addition, the choice of particular measures and their design must follow the existing guidance for each particular practice, i.e. habitat creation, habitat restoration, population reinforcement, species reintroduction, or any other measure considered in the compensatory programme.

5.5.4. *Extent of compensation*

The extent required for the compensatory measures to be effective is directly related to the quantitative and qualitative aspects inherent to the elements of integrity (i.e. including structure and functionality and their role in the overall coherence of the Natura 2000 network) likely to be impaired and to the estimated effectiveness of the measures.

Consequently, compensation ratios are best set on a case-by-case basis and must be initially determined in the light of the information from the Article 6(3) appropriate assessment and ensure ecological functionality. The ratios may then be redefined according to the results observed when monitoring the effectiveness, and the final decision on the proportion of compensation must be justified.

There is wide acknowledgement that ratios should be generally well above 1:1. Thus, compensation ratios of 1:1 or below should only be considered when it is shown that with such an extent the measures will be fully effective in reinstating structure and functionality within a short period of time (e.g. without compromising the preservation of the habitats or the populations of key species likely to be affected by the plan or project nor their conservation objectives).

5.5.5. *Location of compensatory measures*

Compensatory measures should be located in areas where they will be most highest effective in maintaining the overall coherence of the Natura 2000 network. This entails a set of pre-conditions that any compensatory measure should meet:

- The area selected for compensation must be within the same biogeographical region (for sites designated under the Habitats Directive) or within the same range, migration route or wintering area for bird species (i.e. sites designated under the Birds Directive) in the Member State concerned. Furthermore, the area should provide functions comparable to those which had justified selecting the original site, particularly regarding adequate geographical distribution.

- The area selected for compensation must have - or must be able to develop - the specific features attached to the ecological structure and functions, and required by the habitats and species populations. This relates to qualitative aspects like the uniqueness of the assets impaired and requires that local ecological conditions be taken into account.

- Compensatory measures must not jeopardize the preservation of the integrity of any other Natura 2000 site contributing to the overall coherence of the network. When carried out on existing Natura 2000 site(s), the measures must be consistent with the conservation objectives of the site(s) and go above the conservation measures established under Article 6(1). Management plans will be a useful reference to steer sensible compensation measures.

In addition, there is general agreement that the local conditions necessary to reinstate the ecological assets at stake are found as close as possible to the area affected by the plan or project. Therefore, locating compensation within or near the Natura 2000 site concerned where suitable conditions for the measures to be successful seems the most preferred option. However, this is not always possible and a range of priorities should therefore be applied when searching locations that meet the requirements of the Habitats Directive:

1) Compensation within the Natura 2000 site, provided the necessary elements to ensure ecological coherence and network functionality exist within the site.

2) Compensation outside the Natura 2000 site concerned, but within a common topographical or landscape unit, provided the same contribution to the ecological structure and/or network function is feasible. The new location can be in another designated Natura 2000 site or a non-designated location. In the latter case, the location must be designated as a Natura 2000 site and be subject to all the requirements of the Nature Directives.

3) Compensation outside the Natura 2000 site, in a different topographical or landscape unit. The new location can be another designated Natura 2000 site. If compensation takes place on a non-designated location, this location must then be designated as a Natura 2000 site and be subject to all the requirements of the Nature Directives.

New designations forming part of compensation measures must be submitted to the Commission before the measures are implemented and before the carrying out of the project but after its authorisation. The new designations should be made available to the Commission through the established channels and procedures as happens with SCI lists and

SPA classifications, and qualify for designation according to relevant criteria under the Habitats and Birds directives respectively.

Best cooperation and coordination shall be ensured by Member States when dealing with the location of compensatory measures in the frame of transboundary projects.

5.5.6. *Timing of compensation*

Timing the compensatory measures calls for a case-by-case approach. The schedule adopted must provide continuity in the ecological processes essential for maintaining the structure and functions that contribute to the overall coherence of the Natura 2000 network. This requires a tight coordination between the implementation of the plan or project and the implementation of the compensatory measures, and relies on issues such as the time required for habitats to develop and/or for species populations to recover or establish in a given area.

In addition, other factors and processes must also be considered:

- A site must not be irreversibly affected before compensation is in place.
- The result of compensation should be operational at the time the damage occurs on the site concerned. Under certain circumstances where this cannot be fully achieved, overcompensation would be required for the interim losses.
- Time lags might only be admissible when it is ascertained that they would not compromise the objective of 'no net losses' to the overall coherence of the Natura 2000 network.
- Time lags must not be permitted, for example, if they lead to population losses for any species protected on the site under Annex II to the Habitats Directive or Annex I to the Birds Directive; priority species listed in Annex II to the Habitats Directive merit special attention.
- It may be possible to scale down in time compensatory measures, depending whether the significant negative effects are expected to arise in the short, medium or long term.

Specific measures to outweigh interim losses that would occur until the conservation objectives are met may be advisable. All technical, legal or financial provisions needed to implement the compensatory measures must be completed before the plan or project implementation starts, so as to prevent any unforeseen delays that may hinder the effectiveness of the measures.

5.5.7. *Long term implementation*

Compensatory measures require that a sound legal and financial basis for long-term implementation and for the protection, monitoring and maintenance of the sites be secured before impacts on habitats and/or species occur. This could involve:

- Providing for temporary protection, even if the SCI/SPA status is only granted later.

- Applying binding enforcement tools at the national level to ensure the full implementation and effectiveness of compensation (e.g. linked to the EIA Directive, if applicable, or to the Environmental Liability Directive; or linking the plan or project approval to the robustness of the relevant provisions for implementing compensatory measures).

- Applying the necessary legal means in case land or rights purchase is deemed essential for the effective implementation of the measures in line with good practice (e.g. standard procedures for compulsory purchase on grounds of nature conservation).

- Establishing monitoring programmes to ensure that the compensatory measures reach their objective and are maintained over the longer term, and if not, that corrective measures are taken to address this, including objectives, responsible bodies and resource needs, indicators, and requirements for reporting to the Commission. This could be best performed by independent bodies specifically set up for the purpose and in close coordination and cooperation with the Natura 2000 authorities.

5.6. WHO BEARS THE COST OF THE COMPENSATION MEASURES?

It appears logical that, in line with the 'polluter pays' principle, the promoter of a plan or project bears the cost of the compensatory measures. It may include it in the total budget submitted to the public authorities in the event of co-financing. In that regard, EU funds could, for example, co-finance the compensatory measures for transport infrastructure that is part of the TEN (Trans-European Networks) and financed from these funds, provided such financial assistance complies with the objectives, rules and procedures applicable to the EU fund in question.

5.7. INFORMING THE COMMISSION OF THE COMPENSATORY MEASURES

The competent national authorities have to inform the Commission of the compensatory measures adopted. Article 6(4) does not specify the form nor the purpose of this information. However, in order to facilitate the process the Commission has prepared a standard form[65] for supplying it with information under Article 6(4). In any case, it is not the Commission's role either to suggest compensatory measures or to validate them scientifically.

The information should enable the Commission to assess the manner in which the adverse effects are compensated for, so that the elements of integrity contributing to the overall coherence of the Natura 2000 network are maintained in the long term. While the national authorities are only specifically obliged to communicate the compensatory measures adopted, it may also prove necessary to provide certain elements relating to the studied alternative solutions and to the imperative reasons for overriding public interest which have led to the approval of the plan or project, insofar as these elements have affected the choice of the compensatory measures.

[65] Available at: http://ec.europa.eu/environment/nature/natura2000/management/guidance_en.htm

NESTLING DIET AND FISH PREFERENCE OF BITTERNS *BOTAURUS STELLARIS* IN BRITAIN

GILLIAN GILBERT, GLEN TYLER & KEN W. SMITH



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The first assessment of both nestling diet composition and selection of fish prey by Bitterns *Botaurus stellaris* in Britain. We provide quantitative information on the diet of nestling Bitterns, examine the factors influencing diet composition and determine whether adult females are choosing particular species and size of prey. Sixty regurgitate samples from 44 broods were examined during visits to Bittern nests made at nine sites in England from 1996 to 2001. Compositional analysis was used to assess influence of age, season and year effects on diet. The fish component of the diet was compared with species found to be generally available within each site from electro fishing data. Eel *Anguilla anguilla* and Rudd *Scardinius erythrophthalmus* made up the greatest proportion of biomass of the diet and this proportion did not significantly change with the age of the chicks. The amount of Eel in the diet changed during the season and the amount of Rudd between years. From those fish species available, female Bitterns preferred to feed chicks on Nine-spined Sticklebacks *Pungitius pungitius,* Eels (0-40g), Three-spined Sticklebacks *Gasterosteus aculeatus* and Rudd (0-20g). It is likely that female Bitterns feeding young in Britain have a limited prey choice. To optimise food availability we need better understanding of the seasonal habitat requirements of key fish species within reedbed-dominated sites.

Key words: *Botaurus stellaris* – chick diet – prey preference – prey selection

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INTRODUCTION

The Bittern *Botaurus stellaris* is a rare breeding species in Britain and on the red list of UK Birds of Conservation Concern (Gibbons *et al.* 1996). The decline from approximately seventy booming males in the 1950s to twenty by the end of the 1980s (Smith & Tyler 1993) has been directly attributed to loss of habitat and seral succession at those sites that retained birds (Tyler *et al.* 1998). As large areas of wetland were removed, drained, damaged by coypu (Boorman & Fuller 1981) or neglected, wet reedbed, the habitat needed by Bitterns, became very rare (Gilbert *et al*. 1996). Significant effort and resources are being devoted to rehabilitating reedbeds (RSPB 1996; RSPB 2001) and many of these sites have had increases in breeding Bittern numbers (Smith *et al*. 2000). There were thirty booming male Bitterns in Britain in 2001, an increase from eleven in 1997 (Martin *et al*. 2001). Bitterns eat a variety of food (Cramp 1992), but in Britain they were thought to concentrate mainly on fish prey as the numbers and species of amphibians are limited at most Bittern breeding sites. Increased breeding numbers and colonisation of Bitterns has, so far, come mainly from rehabilitation of existing reedbed sites that already have mature fish populations. Electro fishing surveys of fish species and relative density at a variety of sites with and without Bitterns have shown that Rudd *Scardinius erythrophthalmus*, Eel *Anguilla anguilla*, Three-

spined Stickleback *Gasterosteus aculeatus* and Nine-spined Stickleback *Pungitius pungitius* are common to sites occupied by nesting Bitterns (Gilbert & Jose 2003). Fish were found to be at generally low densities in this type of lowland shallow vegetated site, with fewer species at lower density at newly created sites compared with more mature ones (Gilbert & Jose 2003). Whilst declines in fish populations have not necessarily contributed to the historical decline of Bitterns, it is possible that prey availability is limiting the growth of the current population. Preliminary data from visiting Bittern nests in Britain show that chick starvation may contribute to the current low breeding success at existing sites (unpubl. data) and the current lack of breeding at newly created sites is likely to be related to their isolation and a slow build up of an available food supply.

There is little known about Bittern diet. Of six published studies, four describe the stomach contents of dead adult birds, (Madon 1935, nine adult stomachs in France; Moltoni 1948, 17 adult stomachs in Italy; Vasvári 1929, 51 adult stomachs in Hungary; Gonzalez *et al.* 1984, six adult stomachs in Spain) and two describe nestling diet (Gaukler & Kraus 1965, diet of nestlings in 13 German nests and Pchelintez 1990, diet of nestlings from three Russian nests). Of these studies, four ranked fish first by frequency, two ranked amphibians first by frequency. Five ranked fish first by weight, one ranked amphibians first by weight. A variety of fish species was most important in the diet, amphibians commonly occurred, invertebrates made up a small percentage, and birds, mammals and reptiles occurred regularly in small numbers. Adult Bitterns are difficult to observe and any feeding observations are likely to be biased towards those made in unusually open habitats. Small cameras have been used to film female Bitterns at the nest, but food is regurgitated directly to chicks and is not easily identified in this way. Bitterns do not excrete material that provides good dietary information. They do regurgitate a hard bolus of indigested material, however, these may be biased against easily digested prey and can underestimate the fish biomass (Amat & Herrera 1977). The only way of accurately quantifying diet was to sample regurgitates of chicks, as has been done for similar piscivorous bird species (Moser 1986; Fasola *et al.* 1993; Barbraud *et al.* 2001). The diet of nestling Bitterns does not necessarily represent the suite of prey species important through the season to the opportunistic adult Bittern in Britain. It does provide information crucial to advancing our efforts of improving the likelihood of Bittern nesting and increasing their breeding success. This paper examines the composition of Bittern nestling diet, what influences diet composition and whether adult Bitterns select specific fish prey from that generally available.

METHODS

Regurgitate samples from nestlings were examined during visits to Bittern nests made at nine sites in England from 1996 to 2001. For reasons of confidentiality only Minsmere is named, other sites are coded (Table 1). There was discrepancy between sites in the numbers of broods sampled, due to the paucity of nests at most sites. Nest visits were made primarily to ring and radio-tag Bittern nestlings and therefore coincided with days when chicks were estimated to be of the appropriate age. Nest visits were always kept to a minimum and none were made specifically to collect regurgitate samples. Bittern nestlings regurgitated on handling, and once collected, the identity, weight, size and condition of items within each regurgitate were noted. Larger items were placed back on the nest, as video evidence showed that these were re-consumed by chicks and/or the adult female; smaller items were removed for closer examination. The weight of a proportion of fish samples had to be determined from length estimates, as they had been partly digested. Electro fishing surveys specific to the nesting sites provided length/weight relationships for all fish species. Some invertebrates, amphibians, birds and mammals in regurgitates were partly digested. Their weight was estimated based on the aver-

Table 1. The number of Bittern nests with viable chicks visited at each site in each year (N), the number of visits to nests at which young were handled (V), and the resulting number of regurgitates (R). Most of the sites are given a coded identity for reasons of confidentiality. Two RSPB Reserves are named, those are Minsmere in Suffolk and L.M. is Leighton Moss in Lancashire.

	Minsmere			L.M.			Lincs.			Suffolk 2			Suffolk 3			Norfolk 1			Norfolk 2			Broads 1			Broads 2		
Year	N	V	R	N	V	R	N	V	R	N	V	R	N	V	R	N	V	R	N	V	R	N	V	R	N	V	R
1996	2	3	3																								
1997	4	11	8																								
1998	5	19	10	1	3	0				1	2	0				2	4	1				1	2	2			
1999	10	24	12							5	7	5				1	2	1							1	2	0
2000	6	13	4	1	1	0				1	2	1	1	2	1				1	1	0				1	2	0
2001	10	20	11	1	1	0	1	2	1																		
Total	37	90	48	3	5	0	1	2	1	7	11	6	1	2	1	3	6	2	1	1	0	1	2	2	2	4	0

age weight of non-digested items. Fish within regurgitates were categorised according to species and other items to within one of the following categories: shrimp, mammal, bird, amphibian or invertebrate (details in Table 2).

The relative abundance of fish species, biomass and number were sampled at each of the Bittern nesting sites during three winters 1996/1997, 1999/2000, 2000/2001. Experienced, trained personnel conducted all fishing under Environment Agency Section 28 licence. The equipment used was a single anode (40 cm diameter) powered by a 2.5 kVA generator and Millstream control box with pulsed D.C. (50 or 100 Hz) output maintained between 2 and 4 A. The same survey method and equipment was used at each site in each year. Surveys were conducted along transects from a small boat. At least nine transects were completed at each site, with each one being at least 20 m long, the minimum length of transect surveyed at any site was 260m. Bittern feeding habitat was known, from radio tracking, to be in wet reedbed associated with an open water edge (unpubl. data). Transects were distributed within sites as evenly as possible according to the amount of reed/water edge and the number of separate water bodies. Transects were fished along open water immediately adjacent to the reed edge, this included water within more sparse emergent vegetation at the edge. Without being able to electro fish effectively within denser reedbed vegetation, this technique provided a sample of fish species that was as representative as possible of what would be available to Bittern. The sphere of influence of the anode was measured at each site to be greater than 50 cm. The area surveyed was calculated as length x width of transects. All fish stunned were caught, identified, weighed and measured; those fish seen to escape were identified and their size estimated where possible. Fishing did not proceed if wet or windy weather created surface water ripples rendering fish difficult to see. The results were not used to calculate the fish populations at each site, only as a relative measure of fish biomass and species present. Electro fishing was undertaken in the winter for three main reasons: to reduce disturbance during the bird breeding season, to allow better visibility in shallow waters because of fewer macrophytes and to increase catch efficiency because fish are less mobile. Potentially, seasonal movement of fish would render the electro fishing catch composition of a site in winter very different from that in spring or summer. Electro fishing at two sites (Suffolk 2 and Suffolk 3) in spring (end of March) and in winter (late November) in 1999 showed greater biomass of catch for both sites in winter (257% more at Suffolk 2 and 80%

Table 2. Main groups of prey in nestling Bittern diet with overall proportions by biomass, mean proportions per brood and percentage occurrence per brood. The invertebrate group does not contain a full species list only to order level at worst.

	Proportion of biomass		Occurrence (%)
	Overall (n = 574)	Brood mean ± SD (n = 42)	Brood (n=42)
Diet group			
Rudd *Scardinius erythrophthalmus*	0.356	0.296 ± 0.404	48.8
Eel *Anguilla anguilla*	0.298	0.245 ± 0.364	36.6
Amphibian :	0.068	0.124 ± 0.278	21.9
Common frog *Rana temporaria*			
Smooth newt *Triturus vulgaris*			
9 Spined Stickleback *Gasterosteus aculeatus*	0.072	0.123 ± 0.229	36.6
3 Spined Stickleback *Pungitius pungitius*	0.019	0.078 ± 0.227	19.5
Invertebrate :	0.027	0.065 ± 0.117	43.9
Beetles and larvae: *Dytiscidae, Hydrophilidae, Dryopidae, Notonectidae, Chrysomelidae*			
Dragonfly and larvae: *Libellulidae*			
Butterfly and Moth larvae: *Lepidoptera*			
Snail: *Valvatidae, Bithynia tentaculata*			
Spider: *Clubionidae*			
Fly : *Diptera*			
Parasite: *Schistocephalus solidus*			
Leech: *Hirudinae*			
Mammal :	0.111	0.028 ± 0.149	4.9
Water vole *Arvicola terrestris*			
Water shrew *Neomys fodieus*			
Tench *Tinca tinca*	0.042	0.023 ± 0.146	2.4
Common Freshwater Shrimp *Gammarus pulex*	0.003	0.017 ± 0.084	4.9
Bird *Acrocephalus schoenobaenus*	0.004	0.001 ± 0.008	2.4

at Suffolk 3). There was no difference in the catch composition at Suffolk 2, even when fish were categorised by size (the same size categories as in Appendix 1) (MANOVA of log ratio differences: Wilk's $\lambda = 0.92$, $P = 0.31$, $n = 30$). There was a difference in the catch composition at Suffolk 3 (Wilk's $\lambda = 0.89$, $P = 0.04$, $n = 30$). This was due to a significant decrease in the winter proportion of Roach *Rutilus rutilus* in the catch compared to spring, (ANOVA on arcsine transformed proportions: $F_{1, 28} = 4.6$, $P = 0.02$). There was no change in the catch composition for the other species between periods at Suffolk 3. Noble et al. (*in press*) electro fished at Minsmere during the season in 1999 and found no change in the age structure of Rudd associated with the reed margin between periods, although he did find significant movement of Roach away from this habitat as the season progressed. We concluded that electro fishing in the winter may give a poor impression of the number of Roach on site, but should provide as good an estimate of other species as fishing in the spring. Electro fishing catchability varies between species, in particular Eels; which tend to be on or in the substrate, are more difficult to catch than other species such as Rudd. We may

have underestimated the proportion of Eels relative to other species. The data were expressed for each fish species as a proportion of the whole catch for each site within the year fishing took place.

Statistical analyses

Variation was examined by multivariate analysis of variance (MANOVA Tabachnick & Fidell 2001). For all MANOVA tests, we checked the residuals were normally distributed. Where data had a non – normal distribution, randomisation tests were carried out in accordance with Brickle & Harper (1999) (see Acknowledgements). Further investigation of how the variance effected diet composition was done by univariate analysis of variance, testing each dependant variable (diet group) in turn. We adopted Bonferroni - adjusted significance levels (Dunn – Sidak method) (Sokal & Rohlf 1995) of 0.008 and 0.015 which were equivalent to overall significance levels of approximately 0.05 and 0.1 respectively. In general, it was not possible to identify adult females from one year to the next. Therefore, the independence of data from broods in different years was not guaranteed.

Broods were categorised into two age groups, with mean age of less than ten days and greater than ten days. Brood size varied between one and five chicks and three seasonal periods were considered: early (up to 20 May), middle (21 May to 20 June) and late (after 21 June). Variation in the weight of whole regurgitates was examined for broods from all sites and years. Where more than one visit resulted in regurgitates from the same brood, only one (chosen at random) was included in the analysis. We tested for the effect of brood age (two levels), brood size (5 levels), site (six levels), year (six levels) and season (three levels). Compositional analyses (Aitchison 1986; Brickle & Harper 1999; Cummins & O'Halloran 2002) were used to identify factors influencing variation in the relative proportions of the biomass of prey groups in regurgitates. The proportions of prey falling into each category were calculated for each brood, on each visit, after pooling data from all chicks. This involved converting the proportions to log ratios. Log ratios cannot be calculated for proportions equal to zero, in these cases a very small proportion was added (0.0001) (Aebischer *et al.* 1993). Compositional analyses were only done on broods from Minsmere and Suffolk 2 and diet groups were reduced to six: Rudd, Eel, amphibian, stickleback (both nine and three - spined), invertebrates and others (comprising the last four groups on Table 2). The diet group "others" was used as the denominator in the log ratio transformation. The initial dataset comparing diet composition with age of chicks included eight broods, visited twice; both visits produced regurgitates. Seven of the broods were from Minsmere and one from Suffolk 2. The mean age of the brood was less than ten days on the first visit (average age 6.75 days, 4-9 range) and greater than ten days on the second (13.1 days, 11-15 range). The log ratios for the second sample from each brood were subtracted from the corresponding one from the earlier sample and the differences were examined for their deviation from zero. Further analyses included all broods from Minsmere (32 broods) and Suffolk 2 (5 broods), these also split into two age groups of less than ten days (average 6.1 days, 4-8 range) and older than ten days (average 12.7 days, 10-20 range). All analyses of electro fishing catch excluded any fish that weighed more than 300g. Compositional analysis on the electro fishing catch, used data from Roach, Pike *Esox lucius* and Flounder *Platichthys flesus*, in a single group as the denominator. Multivariate analysis of variance was used to look for differences in the catch composition between sites (3 levels: Minsmere, Suffolk 2 and "others") and years (3 levels: 1996/1997, 1999/2000, 2000/2001). The data for each site was as log ratio proportions of caught fish species for individual sections fished (38 Minsmere sections, 22 Suffolk 2 sections, 48 sections from the other sites.

Comparisons between regurgitate and electro fishing catch compositions was made for Minsmere and Suffolk 2 and only concerned the fish component of regurgitates. Two regurgitates from

these sites did not include fish and therefore the number of broods involved in analyses was thirty-five. The larger fish species common in regurgitates, Rudd and Eels, were split into different size classes (Appendix 1). Log-ratio proportions were calculated for fish species and size categories, for both regurgitate and electro fishing samples, using Rudd size 31-40g as denominator. Flounder and Pike were not included as they were absent from nestling diet and only one of each was caught by electro fishing, The log ratio differences were calculated for each bird using the electro fishing sample from the corresponding year and site. Prey items were ranked in order of preference by construction of a matrix of log ratio differences between regurgitate diet and electro fishing catch, following Aebischer & Robertson (1992).

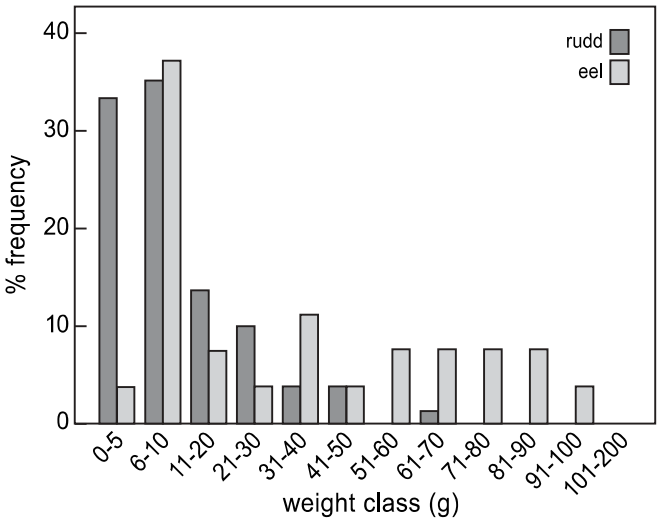


Fig. 1. The percentage frequency distribution of size classes of Rudd *Scardinius erythrophthalmus* and Eels *Anguilla anguilla* found in the regurgitates of Bittern chicks from 42 broods at 6 British sites from 1996-2001.

RESULTS

There were 60 regurgitates produced from 44 of 56 broods visited, at six sites from 1996 to 2001 (Table 1). Fifty-seven of these regurgitates were completely collected and included in diet estimations, these were from 42 broods and contained 574 items (mean = 9.94 ± 17.6 per regurgitate, range 1-91). Of the 42 broods which had regurgitates collected, 32 were from Minsmere, five from Suffolk 2 and only a snapshot of the diet was obtained from the four other sites. The weight of whole regurgitates varied (mean = 46.86g ± 49.87g per regurgitate, range 0.75-268g) between broods, but only brood size was driving this variation (MANOVA: Wilks $\lambda = 0.793$, $P = 0.037$, $n = 42$), there was no effect of year, age, season or site.

Rudd and Eel dominated the biomass of nestling diet at about 65% (Table 2). The contribution of mammals to the overall diet biomass was high (> 11%) because of the relatively large size of Water vole *Arvicola terrestris* prey, but this was an unusual occurrence in the diet compared to invertebrates, amphibians and sticklebacks (Table 2). The size range of individual Rudd in regurgitates was smaller than that of Eels (Fig. 1). The arcsine transformed biomass proportions of all diet groups in Table 2 were compared

Table 3. Seasonal and annual changes in the biomass composition of nestling Bittern regurgitates, for two dietary components, Eel *Anguilla anguilla* and Rudd *Scardinius erythrophthalmus*. Figures in each cell are the percentage biomass of all items of that prey group and in parentheses, the percentage of samples containing that prey group. Only data from 35 broods at Minsmere and Suffolk 2 are included.

	Season			Year					
	early	mid	late	1996	1997	1998	1999	2000	2001
Eel	14 (20)	28 (23)	38 (77)	68 (100)	10 (25)	19 (20)	16 (40)	22 (67)	25 (37)
Rudd	65 (80)	37 (39)	13 (44)	29 (50)	92 (75)	64 (80)	7 (27)	78 (100)	32 (50)

between Minsmere, Suffolk 2 and "other sites" combined into one group. There were more shrimp at Suffolk 2 (ANOVA: $F_{2,39} = 8.85$, $P = 0.001$), but only marginally more Rudd in the diet of Bittern chicks at Minsmere (ANOVA: $F_{2,39} = 3.31$, $P = 0.046$) and more Tench *Tinca tinca* and birds at the other sites (ANOVA: Tench: $F_{2,39} = 4.54$, $P = 0.017$; Bird: $F_{2,39} = 4.3$, $P = 0.014$).

Eight broods which had repeat visits that produced regurgitates were compared, but there was no variation in diet composition with chick age (MANOVA: age, Wilk's $\lambda = 0.86$, $P = 0.35$, tested against 999 randomisations).

Since we found no difference in the composition of diet between the paired samples we pooled the data for each brood. Where regurgitates were collected from the same brood on more than one day, we used the mid – point date. We tested again for the effects of age (2 levels) on the diet composition of all broods from Minsmere and Suffolk 2 and included factors for seasonal periods (3 levels), years (6 levels), brood size (5 levels) and site (2 levels). Multivariate analysis of variance showed significant variation with seasonal periods and years, but no variation with age, site or brood size (season: Wilks $\lambda = 0.96$, $P = 0.037$, $n = 37$; year: Wilks $\lambda = 0.95$, $P = 0.04$, n = 37). The proportion of Rudd in the diet varied significantly between years (ANOVA: $F_{5,31} = 3.9$, $P = 0.006$), with a particularly low proportion in 1999 (Table 3). The proportions of Eel and Rudd varied with season (ANOVA: Eel had a positive effect $F_{2,34} = 4.8$, $P = 0.014$; Rudd had a negative effect $F_{2,34} = 3.38$, $P = 0.048$) (Table 3) although only the effect of Eels was weakly significant.

Eight fish species made up the catch composition: Rudd, Eel, three and nine – spined sticklebacks, Tench, Roach , Pike and Flounder from all sites. There was no difference between years but there was a significant difference between sites (MANOVA: Wilks $\lambda = 0.67$, $P = 0.012$, $n = 108$). There was a significantly greater Eel biomass at Suffolk 2 (ANOVA: $F_{2,105} = 7.83$, $P = 0.001$), than at Minsmere or other sites. At other sites, there was a significantly higher biomass of Tench (ANOVA: $F_{2,105} = 10.22$, $P < 0.001$).

Multivariate analysis of variance showed significant non-random selection of the available fish (MANOVA: Wilk's $\lambda = 0.71$, $F_{28,952} = 15.02$, $P < 0.0001$). Dietary items were ranked in order of their preference (after Aebischer and Robertson 1992) (Appendix 1). The order of preference, when size of item was taken into consideration, was: Nine-spined stickleback > Eels (0-20g) > Three-spined stickleback > Eels (21-30g) > Rudd (6-10g) > Eels (31-40) > Rudd (0-5 & 11-20g) > Rudd (21-200g) > Eel (41-300g) >Tench > Roach.

DISCUSSION

Nestling diet was composed of relatively few prey species, but was predominantly fish as found in previous studies. At the two sites with most nesting activity, Nine-spined sticklebacks, Eels, Three-spined sticklebacks and Rudd were the fish prey of choice, whereas Roach, and Tench were avoided. The reason for the preference is most likely due to the differences in behaviour of the fish species. Rudd and Eels are adapted to penetrating into littoral vegetation (Johansson 1987; Winfield 1986; Tesch 1977), but other fish species native to Britain vary in how much they associate with dense macrophyte or littoral habitat. The limitations of existing electro fishing equipment and techniques did not allow surveys inside dense stands of *Phragmites* where Bitterns predominantly feed. It could be that the fish selected by female Bitterns were the only ones truly available inside wet reedbed. A new design of electro fishing equipment recently developed by the Hull International Fisheries Institute may allow better understanding of the selective use of reedbeds by fish (Noble *et al*. 2002).

The size range of Rudd fed to nestlings was more limited than for Eels, which is perhaps explained by gape size (Moser 1986). Heavier Eels would still be narrow enough to be swallowed, in fact on a couple of occasions chicks were found to have the tail of an Eel still protruding from their bills while the other end was being

digested. We did not detect any differences in composition of prey items fed to broods of different ages, but there did seem to be a change in the diet composition between years and during the season. Although during the season, the proportion of Eel increased and Rudd decreased, there was not an obvious switch from one prey to the other. There may have been a change in the availability of Rudd and Eels, such that Eels became more available, perhaps due to their increased activity in warmer weather (Tesch 1977). There is no evidence that Rudd move out of the wet reedbed margin between May and July, even though this period coincides with spawning activity. Between year changes in diet composition, particularly concerning Rudd, are difficult to explain. Recent studies (Noble et al. *in press*) suggest that only limited age classes of Rudd were present at Minsmere in 1999 suggesting a recruitment problem.

Bitterns in Britain have a limited selection of fish species available to them. They also require that fish populations breed successfully each year to keep providing a healthy range of smaller size class fish. Given that availability may change through the season, or between years, it is important for an adult female, feeding young, to have as many options as possible. The four fish species common to British sites with nesting Bitterns (Gilbert & Jose 2003) were also the four species of preference in the diet. Female Bitterns may be using their availability as one of their cues whether to settle at a site and nest. However, the presence of amphibians, mammals, crustaceans, birds and invertebrates in the diet illustrates the opportunistic nature of foraging Bitterns. Declines in aquatic mammal (Jefferies *et al*. 1989) and amphibian (Houlahan *et al*. 2000) populations have mirrored historic declines in the British Bittern population. With this in mind, management of reedbed sites as healthy wetland mosaics will not only take into consideration the needs of other vulnerable species, but will provide a wider foodbase for Bitterns.

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SAMENVATTING

Het voedsel van nestjongen van de Roerdomp *Botaurus stellaris* werd bekeken aan de hand van voedsel dat werd uitgebraakt tijdens nestbezoeken (60 braaksels van 44 broedsels) aan broedplaatsen in Engeland in 1996-2001. Dit werd vergeleken met het voedselaanbod, zoals vastgesteld met behulp van elektronische visvangst in drie winters in dezelfde periode. Ongeveer 65% van het menu bestond uit Paling *Anguila anguilla* en Ruisvoorn *Rutilus erythrophthalmus*. De samenstelling van het voedsel veranderde niet bij toenemende leeftijd van de jonge Roerdompen, maar het aandeel Paling in het voedsel nam licht significant toe met vorderend seizoen (doordat hun activiteiten toenemen met stijgende temperatuur in de zomer?). Het aandeel Ruisvoorns varieerde significant van seizoen tot seizoen, met vooral in 1999 een klein aandeel. Vergeleken met het visaanbod voederden roerdompvrouwtjes hun jongen het liefst met Tien- *Pungitius pungitius* en Driedoornige Stekelbaarsjes *Gasterosteus aculeatus*, Paling (tot 40 g) en Ruisvoorn (tot 20 g). De beschikbaarheid van deze vissoorten zou een sturende rol kunnen spelen bij de nestplaatskeus van Roerdompen, temeer daar blijkbaar de keus beperkt is. Aan de andere kant, Roerdompen zijn tamelijk opportunistisch in hun voedselkeus. Naast de al genoemde soorten werden ook Zeelt *Tinca tinca*, (zoetwater-) vlokreeftje *Gammarus pulex*, insecten, amfibieën, vogels en zoogdieren uitgebraakt gevonden. (RGB)

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Appendix 1. Simplified ranking matrix for nestling Bittern diet based on comparing fish species and sizes present on site, with those found in regurgitate samples. All size categories are in units of grams. Matrices of logratio differences between regurgitate biomass composition and relative fish biomass on site were constructed for each of 35 broods from Minsmere and Suffolk 2. Each mean element in the matrix was replaced by its sign. A triple sign indicates significant deviation from random at $P < 0.05$, (after Aebischer & Robertson 1992). The size classes of Rudd and Eel are grams. The number of positive values corresponds to the rank, the highest rank number is the more preferred item.

	Rudd 0-5	*Rudd 6-10*	*Rudd 11-20*	*Rudd 21-30*	*Rudd 31-40*	*Rudd 41-50*	*Rudd 51-60*	*Rudd 61-70*	*Rudd 71-80*	*Rudd 81-90*	*Rudd 91-100*	*Rudd 101-200*	*Eel 0-5*	*Eel 6-10*	*Eel 11-20*	*Eel 21-30*	*Eel 31-40*	*Eel 41-50*	*Eel 51-60*	*Eel 61-70*	*Eel 71-80*	*Eel 91-100*	*Eel 101-200*	*Eel 201-300*	*3spine*	*9spine*	*Tench*	*Roach*	*Rank*
Rudd0-5	0	—-	-	+	+	+	+++	+++	+++	+++	+	+++	—-	—-	—-	—-	-	+	+	+	+	+++	+++	+++	—-	—-	+	+++	18
Rudd6-10	+	0	+	+++	+++	+++	+++	+++	+++	+++	+++	+++	-	—-	-	-	+	+++	+++	+++	+++	+++	+++	+++	-	—-	+++	+++	21
Rudd11-20	+	-	0	+	+	+	+++	+++	+++	+++	+	+++	—-	—-	—-	—-	-	+	+++	+	+++	+++	+++	+++	—-	—-	+	+++	19
Rudd21-30	-	—-	-	0	-	-	+++	+++	+	+++	-	+++	—-	—-	—-	—-	-	-	+	+	+	+++	+++	+++	—-	—-	+	+++	13
Rudd31-40	-	—-	-	+	0	-	+++	+++	+++	+++	+	+++	—-	—-	—-	—-	-	+	+	+	+	+++	+++	+++	—-	—-	+	+++	16
Rudd41-50	-	-	-	+	+	0	+++	+++	+	+++	+	+++	—-	—-	—-	—-	-	+	+++	+	+	+++	+++	+++	—-	—-	+	+++	17
Rudd51-60	—-	—-	—-	—-	—-	—-	0	-	—-	—-	—-	+++	—-	—-	—-	—-	—-	—-	-	—-	—-	+++	+++	+++	—-	—-	+++	+++	6
Rudd61-70	—-	—-	—-	—-	—-	—-	+	0	-	-	—-	+++	—-	—-	—-	—-	—-	—-	-	—-	—-	+++	+++	+++	—-	—-	+++	+++	7
Rudd71-80	—-	—-	—-	-	—-	-	+++	+	0	+++	—-	+++	—-	—-	—-	—-	—-	—-	+	—-	—-	+++	+++	+++	—-	—-	+++	+++	10
Rudd81-90	—-	—-	—-	—-	—-	—-	+++	+	—-	0	—-	+++	—-	—-	—-	—-	—-	—-	-	—-	—-	+++	+++	+++	—-	—-	+++	+++	8
Rudd91-100	-	—-	-	+	-	-	+++	+++	+++	+++	0	+++	—-	—-	—-	—-	—-	+	+++	+	+++	+++	+++	+++	—-	—-	+	+++	15
Rudd101-200	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	0	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	+	—-	—-	—-	+	+++	3
Eel0-5	+++	+	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	0	—-	-	—-	+++	+++	+++	+++	+++	+++	+++	+++	+	—-	+++	+++	23
Eel6-10	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	0	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+	-	+++	+++	26
Eel11-20	+++	+	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+	-	0	+++	+++	+++	+++	+++	+++	+++	+++	+++	+	—-	+++	+++	25
Eel21-30	+++	+	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	—-	—-	—-	0	+++	+++	+++	+++	+++	+++	+++	+++	-	—-	+++	+++	22
Eel31-40	+	-	+	+	+	+	+++	+++	+++	+++	+++	+++	—-	—-	—-	—-	0	+++	+++	+++	+++	+++	+++	+++	—-	—-	+++	+++	20
Eel41-50	-	—-	-	+	-	-	+++	+++	+++	+++	-	+++	—-	—-	—-	—-	—-	0	+++	+	+	+++	+++	+++	—-	—-	+++	+++	14
Eel51-60	-	—-	—-	-	—-	—-	+	+	-	+	—-	+++	—-	—-	—-	—-	—-	—-	0	-	-	+++	+++	+++	—-	—-	+	+++	9
Eel61-70	-	—-	-	-	-	-	+++	+++	+++	+++	-	+++	—-	—-	—-	—-	—-	-	+	0	+	+++	+++	+++	—-	—-	+++	+++	12
Eel71-80	-	—-	—-	-	-	-	+++	+++	+++	+++	—-	+++	—-	—-	—-	—-	—-	-	+++	-	0	+++	+++	+++	—-	—-	+	+++	11
Eel91-100	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	+++	—-	—-	—-	—-	—-	—-	—-	—-	—-	0	+++	+++	—-	—-	+	+++	5
Eel101-200	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	0	—-	—-	—-	+	+	2
Eel201-300	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	+++	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	+++	0	—-	—-	+	+++	4
3 spine	+++	+	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	-	-	-	+	+++	+++	+++	+++	+++	+++	+++	+++	0	-	+++	+++	23
9 spine	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+	+++	+++	+++	+++	+++	+++	+++	+++	+++	+++	+	0	+++	+++	27
Tench	-	—-	-	-	-	-	—-	-	-	-	-	-	—-	—-	—-	—-	—-	—-	-	—-	-	-	-	-	—-	—-	0	-	0
Roach	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	—-	-	—-	—-	—-	—-	0	0

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Advice for the management of flightlines and foraging habitats of the barbastelle bat *Barbastella barbastellus*

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Frank Greenaway

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Illustrations

Barbastelle flightlines from Ebernoe Common
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Ebernoe Barbastelle Flightline 2
Ebernoe Barbastelle Flightline 3
Ebernoe Barbastelle Flightlines 4,5 & 6
Ebernoe Barbastelle Flightlines 7 & 8
Barbastelle Flightlines from Ebernoe Common across Butcherlands Farm
Woodland flyways
Examples of good and still developing connective cover on barbastelle flightlines
A lost flightline?
Examples of ‘en route’ forage areas
Final forage areas of female barbastelles

1. Understanding barbastelle survey information and records

Recent in-depth radio-tracking and survey work with a number of bat species has begun to show a substantial element of resource partitioning between the sexes in the summer months. There are strong indications that this is the case with barbastelles. A few males may be found within the area of prime habitat used by a maternity colony, but the majority of males are spread far and wide in sub-optimal habitats. There is no discernable sex difference in the echolocation calls of barbastelles. This means that bat detector records of barbastelle calls may represent males who may be several tens of kilometres away from the nearest nursery colony. To understanding the significance of any record of barbastelles, it is essential to know the sex and sexual state of the individual bat concerned.

Until the mid-1990s virtually nothing was known about the habitat requirements of barbastelle bats, but since then, there has been a number of studies across their range in Europe and the UK. Most of what is known about the flightlines and foraging habitats of female barbastelle bats is based on data on a colony in West Sussex that has been studied since 1996. This colony, at Ebernoe Common, was the first in Britain to be studied systematically, and is currently the only one for which there are comprehensive data on the foraging behaviour of multiple females. However, information that is gradually accumulating from other studies supports the evidence presented in this report, which is that female barbastelles make use of well-established, extensive flightlines and foraging habitats that are far removed from their roosting area. Consequently, while the picture that is emerging of their ecology, behaviour and habitat requirements comes largely from one local population, there is good reason to believe that it has much broader applicability. Similar habitat requirements have been observed in Europe.

1.1 Flightlines and foraging habitats of female barbastelles

Female barbastelles congregate in the spring to form colonies in which they give birth and raise their young. Most of these maternity colonies are located in areas of mature woodland, and each colony uses the same area year after year. Every night during the spring and summer females disperse from the woodland along established routes or flightlines to foraging areas that may be several kilometres away. To a large extent a female's ability to feed herself and her offspring will depend on the condition of these flightlines and foraging habitats. Thus, any habitat management plan for barbastelles must take into consideration both the wood in which their roosts are located and the broader landscape surrounding that wood. More specifically, management of the surrounding area must focus on the maintenance of established flightlines and foraging areas.

Patches of old growth oak woodland and the small floodplains of woodland streams and rivers are frequently the habitat types associated with the 'en route' foraging areas. They can be as little as one hectare in extent or, as is often the case with the woodland types, much larger.

3.6 Density of foraging barbastelle bats in the landscape

Foraging barbastelles are spread very thinly in the landscape. Even within the healthy colony from the Ebernoe roost the density of bats in late summer is rather less than one female or juvenile bat to six square kilometres. This figure, of course, includes a very large area of land that the bats very seldom, if ever, use as it consists of unsuitable, largely arable habitats. The foraging habitats themselves form a select and vulnerable set of the more stable and productive habitats; a small percentage of the total area, but rich in diversity.

3.7 Changes in the location of principal foraging territories over time

Drainage of fields and meadows, scrub clearance and the felling or planting of new areas of woodland can all lead to semi-permanent changes in available foraging areas. Such changes have the potential to cause dramatic alteration in the breeding success of barbastelle colonies. When too many foraging opportunities are lost along a flightline, the bats will cease to use it, even if its connectivity is maintained. Increasing foraging opportunities along the sections of flightline used in early evening lowers the requirement for long distance commuting in poor weather. In fine weather the bats normally ignore such close sites and head rapidly for the highly productive distant zones.

3.8 Management of foraging habitats

Management of habitats to promote insect, and especially micro-lepidopteran, biomass is the rather difficult goal. Old established unfertilised grasslands and water meadows are naturally highly productive of insect biomass. Where such areas are exposed to prevailing wind the number of nights when the moths are on the wing can be greatly restricted. Shelter belts or even a series of isolated trees, such as spreading willows, can be highly effective in slowing wind speeds and increasing insect availability.

It is very difficult to make all encompassing statements when dealing with so many different small moth species, however the larvae of many of the more abundant species feed on basal stems and roots rather than the upper sections of grasses. It would seem therefore that light grazing would have little effect on the moth biomass. Hay cutting on the other hand does greatly and suddenly alter local insect availability at a very susceptible time of year for the pregnant bats.

In the scrub and wooded habitats, apart from total clear fell, scrub or understorey removal is potentially the most damaging operation. Understorey plants are the larval foods of many small moths, the Geometridae in particular. Such operations will also increase local wind speeds and lower the humidity, both changes that will lead to lower insect abundance and availability. Woodland that is basically old-growth, but with a very wide selection of types within it, from glade to dense high forest, should be a management aim. Where coppicing work is necessary it should be carried out in small patches so as to maintain woodland humidity and retain low wind speeds.

Forestry operations within foraging areas in woodland adjoining watercourses are another problem. The best that can be done when such work is unavoidable is felling selectively over time, rather than the more usual practice of clear felling. Tree lines should always be left along the watercourse itself. If light, selective felling is well practiced leaving an understorey intact, it can often be beneficial in promoting insect availability. Trees with rot cavities or woodpecker holes should always be left intact.

The small floodplains along streams in woodlands should be left to develop naturally with no attempts to drain them or clear blockages. Open areas that flood in winter on such woodland streams become highly favoured foraging spots in summer.

3.9 Management of land adjoining foraging areas

Generally speaking, barbastelles prefer established natural or semi-natural habitats that are fairly stable in character and require little maintenance. Two external factors do however have a noticeable effect the productivity of barbastelle foraging areas.

Many foraging zones benefit from the drift of insects in from adjoining fields or open habitats. When the open areas themselves are too exposed or low in productivity to be beneficial foraging sites a concentration of insects can still occur on the lee side of the first obstruction to the wind. Such situations are extensively exploited by many bat species including barbastelles. Inherently rich old oak woodland bordered by old unimproved grasslands or pasture is the ideal situation for this, but it does also occur on hedgerows across open ground.

Arable land adjoining woodland or wet foraging habitats has the constant possibility of contamination damage from the run off of herbicides, pesticides or fertilizer, any or all of which will alter availability or abundance of insect life. Contamination of the watercourse itself may cause total loss of foraging areas where exploited species are moths with aquatic larval stages.

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Report Number
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Woodland management advice for Bechstein's bat and barbastelle bat

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English Nature Research Reports

Number 658

Woodland management advice for Bechstein's bat and barbastelle bat

Frank Greenaway
David Hill
July 2004



Barbastelle bat, *Barbastella barbastellus*.



Bechstein's bat, *Myotis bechsteinii*

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1. Introduction

Barbastelle and Bechstein's bats are at the northern edge of their range in Britain with the barbastelle reaching a bit further north than Bechstein's bat. Both species are considered rare in Britain although the Bechstein's bat has at least the potential to have high local populations in large areas of dense deciduous woodland. The barbastelle on the other hand has small populations spread over very large areas at all its known colonies. Both animals are extremely hard to locate and observe and viable populations can remain unnoticed even in well-studied woodlands.

As with several other bat species, the distribution of nursery colonies of both of these animals seems to be in the prime older woodland habitat locations. Frequently a scatter of lone male animals is recorded from across the wide areas of intervening space between these nursery colonies. These male bats, living in sub-optimal habitats and occasionally using underground roosts in winter, form the bulk of the species records as the tree roosting females in their old undisturbed woodlands very rarely come into contact with people. In consequence the mapped range of both species may considerably misrepresent the range and in particular the colony locations of the species.

Both Bechstein's bat and barbastelle bat will share roosting woodlands as there is some overlap in the structural requirement of the woodlands, albeit for different reasons. Any similarity stops there however as both the types of roost trees they require and the summer foraging habitat requirement are completely different. For the female Bechstein's bat the woodland is both a provider of roosts and the source of forage. For the female Barbastelle, the roost woodland is the communal roost site but the major summer forage habitats can be many miles distant.

Bechstein's bat is a warm temperate deciduous woodland species. It should respond well to the predicted warmer summers and wetter winters global warming will bring to England. Barbastelle bat, also on the northern edge of its range, is arguably already more suited to the maritime climate and has a rather wider distribution reaching further north than Bechstein's bat. Provided suitable habitat exists, global warming should allow the barbastelle to spread further north. Because of its large territorial and multiple foraging habitat requirements barbastelle colonies currently remaining in southern England may come under further pressure from human land use.

2. Bechstein's bat

2.1 The size and structural requirement of woodlands with Bechstein's bat nursery colonies

The ideal Bechstein's bat woodland is an unevenly aged, deciduous woodland with a high number of oaks in the species mix. The woodland would be of a minimum of 40 to 50 hectares in extent and be semi-natural or ancient woodland with a dense mixed species understorey. Very large continuous areas of high forest seem to produce lower numbers and more widely spaced nursery colonies than slightly fragmented structurally diverse forests. Some blocks of continuous high forest may have populations of Bechstein's bats that are almost exclusively males. Similar all male or non-breeding populations can be found in woodlands that are too small to support nursery colonies. The reasons for this resource

of suitable roost trees into the distant future. This requires linked canopy cover with understorey over an area of about 50 hectares with further areas going into canopy decline and others not yet in canopy closure or in sapling stage. The current trend in forestry practice towards a wider remit of wildlife and recreation as well as timber production gives some scope for management practice to improve matters. A forestry timber extraction policy that follows the slow removal of prime individual trees on a continuous basis, rather than clear fell, will avoid sudden crashes in colony population sizes by maintaining adequate canopy cover for foraging.

Improvements in plantation management should include:

1. Creating non-intervention strips along all watercourses within the woodlands. This should include all the small floodplains and steep banks along the woodland streams.
2. Harvesting hardwood trees in plantations only when unavoidable and then by selected felling only, done on a slow continuing basis cutting only the best sound mature timber at appropriate times of the year.
3. Monitoring stands of trees used as nest sites by woodpeckers and leaving these stands as non-intervention until their natural decay.
4. Creating a series of suitable areas within which Green Woodpeckers can forage for ants. These areas should be over and above the woodland area required by the bats to forage in.
5. Ensuring, by new planting if necessary, that all hardwood blocks in nursery colony areas have deciduous woodland connections.
6. Leaving not only hollow trees but the immediate stand of trees around them together with the understorey during any felling operations.

2.11 Artificial roost sites for Bechstein's bats

Nursery colonies of Bechstein's bats will readily colonise suitable bat boxes within their colony territory. The preferred boxes are Schwegler 2fn specials but several other types have been used successfully as well. All these boxes are susceptible to having the entrances blocked by birds nesting material and need constant inspection and cleaning to remain of use. The boxes are also attractive to most other deep cavity roosting bat species, raising the issue that the balance of bat species within a woodland will be disturbed by this use and thus have a potential negative impact on the Bechstein's bats. Whilst bat boxes certainly have a function in the short term to counter a failing habitat they should not be seen as a permanent solution to shortcomings in the availability of suitable trees.

3. Barbastelle bat

3.1 The size and structural requirement of woodlands around barbastelle bats nursery roosts

Barbastelle bat nursery colonies have a strong tendency to be sited towards the north or northwestern side of the brows of quite low wooded hills. The woodland itself is usually semi-natural in character, often ancient in origin and frequently of high natural humidity. High numbers of oaks killed by crown competition or storm damage are often present. A

dense understorey that will aid humidity retention around roost trees and lower wind speeds is frequently present, particularly when the colony is located in areas with naturally low summer humidity. The classic location for a nursery colony is within woodlands on the headwaters of small rivers with the river giving access to the floodplain for summer foraging. Woodlands do not individually need to be particularly large but the connected woodland total needs to be a fair percentage of the landscape. Approximately thirty percent woodland seems to be average within the majority of colonies. A few colonies are known from more open areas with lower woodland cover percentages but such colonies have not been studied on a landscape scale and it is not known how they behave, particularly with respect to individual forage areas or for foraging in the winter months. Such open country colonies are considerably more scattered than those in the preferred heavily wooded country.

Roosts are usually located within a restricted area of the selected nursery woodland block, a 'loyalty area'. The bats move roost around this area over the summer utilising subtle differences in the woodland structure to maximise the benefits of buffering large swings in ambient temperature and humidity.

3.2 Nursery roost site requirements of barbastelle bat

The tree nursery roosts of barbastelle bats are located behind loose plates of bark on dead trees and in narrow cracks and fissures of tall broken tree stumps. Storm damage cracks in the major limbs and stems of large living trees are also used. In western areas loose ivy also seems important. The selected sites are usually very narrow but the bats themselves may be quite close to the surface, frequently visible from the ground with a torch. Protection from the prevailing weather conditions is minimal and colonies move roost with great frequency, often daily when sufficient roost sites are available. The entire nursery colony only comes together into one roost occasionally and is normally in several subgroups scattered across the roosting area of woodland.

When woodland containing suitable trees is no longer available, or when conditions become exceedingly dry, barbastelle bats will switch to other high humidity roost alternatives. An example of this use of alternative sites is the use of a massively timbered stone barn with a thatched roof and deep narrow recesses in the structure used by one colony in East Anglia. All the evidence to date suggests that loyalty to a nursery colony is absolute, female barbastelle bats do not change colonies.

Commercial forestry woodlands are rarely of much use to barbastelles as nursery woodlands. Damaged or dead trees are removed too quickly and the understorey is rarely allowed to develop sufficiently to retain woodland humidity. It is probable that when management for barbastelles nursery roosts in plantation forestry is required it should be tailored to each particular situation as it arises.

3.3 Barbastelle bat roost site requirements at other times of year

Barbastelle bats become less social outside the nursery colony season (May to Sept). When the requirement for very high levels of energy input falls after the young are weaned, the bats reduce the long commuting flights and forage within a few kilometres of the current roost. By October individual bats roosts are much more scattered although it is not unusual for almost exclusively juvenile collective roosts to remain in the nursery woodlands until quite late in the autumn. A number of adults from a colony certainly remain in the same nursery roost area

that has been occupied all summer but do so in a solitary fashion. Many females also disperse outwards towards their individual summer forage areas, especially so if this was a woodland habitat.

Roosts selected are at lower levels and deeper in dense woodland but still retain the typical barbastelle characteristics of rather exposed sites behind loose bark, in tree cracks, or in cold weather conditions, deep in hollow trees. Occasionally individuals will utilise cold buildings through the mid-winter period and more extensively in periods of extreme cold. Prevailing weather conditions have a marked effect on site selection, moderate frost protection normally being given by large areas of dense vegetation around the chosen roost. In spring a slow movement commences back towards the nursery roost area finally culminating in colony roosts reforming in early to late May.

3.4 Foraging habitat requirements of juvenile barbastelle bats

Juvenile barbastelle bats become volant later in the year than most other species, often in early August. The bats are not weaned when they first start to fly and are still incapable of making the long commuting flights to follow their mothers. Radio tracking data suggests that the juveniles initially fly around the immediate area of the roosts and then slowly over a period of weeks follow their mothers further and further towards her forage grounds. During this period they become weaned and start to hunt for themselves. The availability of abundant productive forage close to the roost in this period is a major key to the success of any bat colony. As with all bat species barbastelles are particularly vulnerable to predation in the first few weeks so cover is of equal importance to suitable small insect abundance. Examples of such forage areas are small woodland floodplains and ponds or small river systems with a plentiful shrubby growth of species like willows. These forage areas need to be on the adult female bats flightlines so that she will find and feed her offspring on her return path to the roost woodlands. The critical period is actually quite short for each young bat, perhaps no more than three weeks. However, as the young of barbastelles are normally born spread out over a longer period than other bats the total period with weaning juveniles from a colony can total nearly two months. By the end of each individuals three week weaning period most juveniles are capable of following their parent the full commuting distance and start to wander in what appears from radio tracked animals to be a rather haphazard fashion around the whole of the colonies territory. In practice this apparently random behaviour may actually be a structured following of other adult bats but this is rather difficult to prove. Certainly some of the areas visited by tagged juveniles are within the areas utilised for forage by other adult females.

3.5 Ensuring a continuing supply of roost sites for barbastelle bat colonies into the future

The slowly changing natural structure of the woodland around barbastelle nursery roosts, will over time, slowly cause a drift in the bats ‘loyalty area’ to occur that takes into account the changing nature of the woodlands and their roost availability. It would be a very unusual woodland structure that could continue to produce a range of potential barbastelle roosts over several tens of years. However, unless the nursery roosting woodlands can supply such a flow of suitable trees and cover over these time periods such roost movements will eventually take the nursery roosts beyond the boundaries of reserves set up especially for the bats. Restricting this movement by ensuring a suitable succession of trees and cover should be a major focus of long term woodland management for nursery colonies of barbastelle bats. Woodland

planning and management for this eventuality will require the detailed management of much larger areas of reserve woodlands than may actually be in use as loyalty areas in any given year. The size of this larger area will depend greatly on the diversity of the surrounding woodlands but will need to be in the order of five to ten times the loyalty area in order to ensure minimal movement over a period of one hundred years. Within this wider area, woodland management will need to encourage a succession of tree age structures, over patches corresponding in size to the original loyalty area of the site. Ideally these patches would flow together in such a way that a smooth movement of the colony's loyalty area can occur as conditions alter with time.

3.6 Artificial roost sites for barbastelle bats

Barbastelles are crevice dwelling bats and do not use conventional bat boxes. Special maintenance free boxes designed to offer narrow gaps with a long access length appear much more attractive to them. Boxes like these have been used by barbastelle nursery colonies in summer and lone individuals right through the seasons at a site in West Sussex. Old carpet tiles applied to dead tree trunks like loose bark have also been used successfully in the same area. Rather more exotic but potentially interesting roost creation ideas have included the use of explosives on tree stems and deep slots made with chainsaws into standing tree trunks.

As with artificial roost sites for Bechstein's bat the provision of such structures carries the risk of altering the population structure and species balance of bats in a woodland. This alteration has the potential for unpredicted negative effects on the species that it was hoped to encourage. Winter active bats such as pipistrelles and barbastelles both forage close to their roosts, and may use similar roosts in winter. Altering the conditions within a restricted area of woodland could benefit generalist foraging species such as pipistrelles, which are then likely to out-compete the specialist species such as barbastelles, which rely on woodland foraging close to their roosts in winter. Similar problems may also occur when young barbastelles are first learning to forage for themselves. Artificial structures are by their nature temporary and should only ever be considered when an overriding reason for their use is clearly apparent.

4. Combined woodland management for barbastelle and Bechstein's bats

The two species will roost within one woodland but select physically different roost cavities so competition for roosts between the species will be almost non-existent. The preferred summer foraging habitats of adults from the two species are also well separated and foraging style between the species is sufficiently great to avoid any direct competition.

A potential exists for some prey availability reduction for juvenile barbastelles as they become volant in July or August. It is not considered that this is of any consequence as the preferred prey types are rather different and it is the time of greatest prey abundance.

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A review of British mammals:

population estimates and conservation status of British mammals other than cetaceans

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Executive summary

1. This review covers 64 species and one sub species of terrestrial mammal known, or believed, to breed in Britain. It includes those feral species that have persisted as breeding populations for at least fifteen years, but excludes the cetaceans.

2. For each species there is an assessment of the current status, historical and recent changes in numbers, population trends and population threats. For most species these assessments are based on subjective rather than objective criteria because there are few species for which there are long-term population data.

3. For each species except Nathusius' pipistrelle a pre-breeding population estimate was calculated to provide a base-line against which to measure future changes. Estimates are provided for Great Britain as a whole and separately for England, Scotland and Wales. The results are summarised in Table 14. The minimum aim was to achieve a population estimate with an accuracy to within an order of magnitude, but most are thought to be more accurate. A code is used to identify the level of confidence for each estimate. For ten species the estimate was graded 1 (most reliable), for ten it was graded 2, for 20 it was graded 3, for 19 it was graded 4, and for five the estimate was graded 5.

4. Several problems were highlighted in the course of making these estimates. First, there are very few species for which an estimate of total population size was available; for most, population size was calculated either by estimating their abundance relative to other species and/or by multiplying the amount of suitable habitat by known population density estimates for those habitats. However, even this approach proved problematical for many species, either because density estimates were not available from Britain, or because these estimates were only available for a limited number of habitat types. Furthermore, most population sizes calculated in this way will tend to be over-estimates, because density estimates so derived are invariably based on a limited number of studies in some of the more suitable habitats for that particular species.

5. The review highlights the paucity of population data for many species of mammal, and density data are few or non-existent even for a number of common and/or widespread species such as the hedgehog, house mouse and common rat. Thus, further field studies are needed to improve our knowledge of the distribution and density of most species; only then will it be possible to refine many of the population estimates.

6. Absolute numbers of mammals are perhaps less important than trends in population size and degree of population fragmentation. The known or believed changes in British mammal populations over the last thirty years are summarised: two species have become extinct; eight are known, or believed, to have undergone substantial increases in range and/or numbers; nine have undergone some increase; for 23, population size was believed to have remained approximately stable; nine have undergone small declines in range and/or numbers; and for 14 there have been substantial declines. The species known, or believed, to be increasing in numbers include several already, or potentially, damaging to agriculture or silviculture, such as the rabbit, red deer, sika deer, roe deer and muntjac and several species which are of conservation concern and which previously had been reduced to low levels, e.g. otter and polecat.

7. The population estimates for mammals are compared to those for other vertebrates. Few species of mammal are as rare as the species of bird listed on Schedule 1 of the Wildlife and Countryside Act 1981. However, mammals are less mobile, and minimum viable populations are likely to be larger. The commonest species of

mammal have population sizes an order of magnitude larger than those of the commonest species of bird; the same relationship applies for the rarest species of mammals and birds.

8. There are a number of population threats faced by British mammals. Of the 65 mammals included in the review, seven are known, or believed, to be threatened by competitors, seven by climate change and/or adverse weather conditions, four by disease, seven by population fragmentation or isolation, 31 by habitat changes, five by inter-breeding, 18 by deliberate killing, 25 by pesticides, pollution or poisoning, four by predation and seven by road deaths.

9. The conservation status and legal protection afforded to all species of mammal in Britain are summarised. See Table 15.

10. The conservation status of each species is discussed from a European perspective. On this basis, most of the species of mammal that are rare in Britain have larger populations in Europe. The insectivores are generally about as common in Britain as in the rest of their European range; other than the Bechstein's and barbastelle bats, which are rare throughout Europe, the British populations of bats contribute only a small proportion of the European population; of the lagomorphs, the brown hare and rabbit populations are important in a European context; of the rodents, the grey squirrel and field vole populations constitute a significant proportion of the total European population; of the carnivores and pinnipeds, the otter, badger, common seal and grey seal populations are important in a European context; of the artiodactyls, the red, sika, fallow, muntjac and Chinese water deer populations are important and the populations of Soay sheep and wild goats that are of ancient origin are of particular interest because the populations in Britain are unique and constitute about half the ancient feral caprines in Europe.

11. The monitoring of endangered wild mammal populations is now a statutory responsibility under European Union legislation. This review identifies those species of particular conservation concern from both a British and a European perspective and provides a basis on which to develop a comprehensive monitoring scheme for British mammals.

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Introduction

There are few data on population sizes and/or population trends for most British mammals, although the National Game Bag Census data provide an indication of population trends for some species (Tapper 1992). For many others, however, even such basic data as population densities are absent for more than a few habitats (see review in Corbet & Harris 1991). Detailed surveys have been undertaken for a few species, e.g. badger *Meles meles* (Cresswell, Harris & Jefferies 1990) and wildcat *Felis silvestris* (Easterbee, Hepburn & Jefferies 1991), and for others there are periodic monitoring exercises, e.g. otter *Lutra lutra* (Andrews & Crawford 1986; Green & Green 1987; Strachan *et al*. 1990), or annual counts over much of their range, e.g. red deer *Cervus elaphus*; see Clutton-Brock & Albon (1989) for an analysis of the data collected for red deer. Overall, however, unlike the regular monitoring of bird populations (e.g. Stroud & Glue 1991), there is no comprehensive monitoring scheme for British mammals. Neither has there been any attempt to estimate the population size for most species of British mammal, unlike the British avifauna (Morris 1993a), although the status of some of the rarer species has now been reviewed (Morris 1993b). Thus for many species of mammal substantial population changes could go unrecorded, as has occurred for the water vole *Arvicola terrestris* (Strachan & Jefferies 1993) and may have occurred for the hedgehog *Erinaceus europaeus* (P.A. Morris unpubl.). Whilst there are data on the number of hedgehogs killed by gamekeepers, these are not suitable for monitoring changes in hedgehog populations (Tapper 1992). This comparative lack of information on British mammal populations is largely due to the difficulties of counting mammals. For birds, counts are often based on habitat type (e.g. gardens), and hence include large numbers of species, whereas for mammals, survey or census techniques need to be tailored to suit the ecology of individual species or, more rarely, groups of species.

Despite the hitherto rather fragmented approach, there is now an increasing amount of information available on the status of many British mammals and, for a small number, good population estimates are available. This review attempts to assess the abundance of all British mammals, to show where deficiencies in knowledge exist, to highlight their status and population trends, and to comment on any potential population threats or other factors that may affect future population size. Finally, the importance of the British mammal fauna is assessed in a European context.

Methods

Basis of the calculations

This review covers all 63 species of native and introduced mammal that are known to have bred in Great Britain in the last thirty years and one species that may have bred in Britain; it also includes one sub species for which there are data on population size, but excludes the cetaceans. The review covers an area of 230,367 km^2, and includes off-shore islands but not the Channel Islands and Isle of Man (Table 1). Records for vagrant or migratory species, and those not known, or not believed, to have bred in Britain, are also summarised. However, introduced species that have not persisted as breeding populations for at least 15 years, such as Mongolian gerbil *Meriones unguiculatus*, golden hamster *Mesocricetus auratus*, raccoon *Procyon lotor* and Himalayan porcupine *Hystrix brachyura* (Smallshire & Davey 1989; Baker 1990), have been excluded. The selection of species included in the review was largely based on the third edition of *The handbook of British mammals* (Corbet & Harris 1991). Where historical records are quoted, the county name used is that given in the original report, and these have not been adjusted to take account of any changes in county names.

Data have been collated from published sources or from specialists for each species, with the minimum goal of producing an 'order of magnitude' population estimate. For many species, there are significant inter-seasonal or inter-annual changes in population size and so, for comparisons between species, we have estimated the size of the overwintering or pre-breeding population. For some species, where accurate population estimates were already available, minimum/maximum figures are given or, where appropriate, estimates with 95% confidence limits. Where no published data on population size were available, for some species these have been provided by key workers. For species where no other population estimates were available, population densities and/or home range sizes for different habitat types were extracted from the literature or supplied by workers on that species, and were used to estimate the total population size, based on the distribution of the species and the area of suitable habitats available. As a general rule, only data from mainland Britain were used for these calculations, since information from the rest of Europe, where there are likely to be differences in both habitat and faunal composition, are unlikely to be representative of Britain.

To estimate habitat availability, data on 42 main habitat types in each of the 32 land classes devised by the Institute of Terrestrial Ecology (Bunce, Barr & Whittaker 1981a, 1981b) were used. The distribution of land classes within England, Scotland and Wales is shown in Table 2. When calculating population size of some species, four land class groups, rather than individual land classes, were used. Thus land classes 2, 3, 4, 9, 11, 12, 14, 25 and 26 formed the arable land class group, land classes 1, 5, 6, 7, 8, 10, 13, 15, 16 and 27 formed the pastoral land class group, land classes 17, 18, 19, 20, 28 and 31 formed the marginal upland land class group and land classes 21, 22, 23, 24, 29, 30 and 32 formed the upland land class group (Barr *et al*. 1993). The habitat data were collected by surveying 2455 1 x 1 km squares during the period November 1985 to February 1988 as part of a national badger survey; full details of the habitat data recorded are given in Cresswell, Harris & Jefferies (1990). In addition, a further 165 1 x 1 km squares from land classes on the Scottish islands, not included in the original badger survey, were subsequently surveyed for habitat data only, to ensure that the habitat data were not biased due to the exclusion of islands uninhabited by badgers. For the habitat data, all areas over 0·5 ha or linear features over 50 m long were included. For each land class, mean areas of each habitat type were calculated, excluding areas of sea. Details of the area of the main habitat types in the area covered by this review are given in Table 3. For aquatic species such as water voles, mink *Mustela vison* and otters, data on abundance were available for water authority regions, and so figures for the lengths of riparian habitats in each authority region are given in Table 4. Data on the rates of habitat change in Britain are given by Barr *et al*. (1993).

When the calculations included in this review were undertaken, there were no habitat data available for Britain as a whole, although there were some data on the land-use changes in

England (Sinclair 1992). Subsequent to the calculations being completed, a report was published on the pattern of land use in Britain in 1990 (Barr *et al*. 1993). This presented land cover data from two different sources: satellite imagery and a very detailed survey of 508 1 x 1 km squares, stratified by land class, with a greater number of squares surveyed within larger land classes. Habitat classifications for these two surveys differed slightly from those used here, and so direct comparisons are difficult. However, the broad results are very similar. For example, in this survey it was estimated that there were 528,000 km of hedgerow, compared to 549,000 km estimated by Barr *et al*. (1993). Thus the habitat data used here provide a reliable basis for estimating the size of the mammal populations in Britain.

Problems with producing the population estimates

There are very few, if any, species for which the population estimates presented here could not be improved. Counting mammals is notoriously difficult, and even for well-monitored species there are uncertainties inherent in the estimates. For example, for the seals, there will be uncertainties about what proportion of the population is hauled-out when the counts are undertaken. Some of the weakest population estimates are those based on population density figures and the area of suitable habitat available. The problem with such an approach is that inevitably there are few density estimates, and those are for a limited range of habitat types, even for the commonest species. Since most workers base their study of a particular species in an area where that species is common and hence easiest to catch, it is difficult to know how typical such density estimates are for the rest of that species' range. For this reason, population estimates using this approach were generally based on density estimates slightly below those reported in autecological studies.

Of particular difficulty in this respect are all the small mammals, the bats and the deer. Hence for the small mammals and the bats, the relative proportions of the different species in a variety of samples were used as a cross-check to determine if at least the relative size of the population estimates were correct, and that the rank order of abundance for the species was appropriate. Thus for small mammals, data were collated from a variety of pellet analyses (Table 5), bottle studies (Table 6) and trapping studies (Table 7); the relative proportions from these studies are summarised in Table 8. These tables are not comprehensive in that they deliberately do not include all the data that were available, since these were strongly biased to southern England, and there were few samples from Scotland or Wales. Thus the samples included in these tables were designed to provide as wide a coverage as was possible, both geographically and by habitat, without excessively biasing the data to a few habitats in southern England. For species such as the water shrew *Neomys fodiens*, for which few, if any, density estimates were available, population size could only be calculated by comparing abundance relative to a common species in a variety of samples from different habitats; in the case of the water shrew, abundance relative to common shrew *Sorex araneus* and pygmy shrew *Sorex minutus* was used. For the bats, few density estimates were available. For greater horseshoe bat *Rhinolophus ferrumequinum* and lesser horseshoe bat *Rhinolophus hipposideros*, total population size was reasonably well known, but for most others there were few data on which to base a population estimate. Thus relative abundance was the only approach available for calculating population size for most bats, and the estimates had to be based on the subjective assessment of relative abundance by experienced bat workers.

For the deer, with the exception of red and to some extent sika *Cervus nippon*, there were few density estimates because counting deer, even in a small area, is notoriously difficult. Most deer census figures tend to be an under-estimate (Springthorpe & Myhill 1985), and using these to calculate total population size is therefore inherently unreliable. The problem is compounded for those species where the latest distribution maps (Arnold 1993) were used to provide an estimate of the occupied range, since these maps provided at best only the minimum distribution for a species. For instance, a survey into the distribution of muntjac *Muntiacus reevesi* increased the number of 10 x 10 km square records from 417 (Arnold 1993) to 745 (Chapman, Harris & Stanford 1994), an increase of 79%. Whilst muntjac may be an extreme example, in that they were still expanding their range, even species with

established ranges were likely to have been under-recorded, and probably by a significant amount.

For each species, a separate population estimate was also calculated for each of England, Scotland and Wales. Where there were regional counts, e.g. for the seals and some of the deer, the three estimates were based on these counts. Where the estimate was based on the area of suitable habitat and known population densities, the figures were based on the availability of suitable habitats in each country. For total population estimates based on abundance relative to other species, e.g. many of the bats, the estimates for each country were based on the proportion of the 10 x 10 km square records in each of the three countries (Arnold 1993), excluding the Channel Islands and the Isle of Man. There are problems with each approach. For instance, the quality of counts from different regions often varied. When using density estimates, it had to be assumed that the same population density occurred in similar habitats throughout the range. Using the proportion of 10 x 10 km square records in different parts of a species' range of necessity placed equal weight on squares with one record and those with many. Thus these regional figures are intended only to give an indication of the relative status of a species in different regions of the British Isles.

Since there are limitations in the data used to produce any of the estimates in this review, they must be viewed as the first stage in a process of estimating the population size for British mammals, and in monitoring population changes and trends. With better data on the distribution and ecology of each species, the estimates for most, if not all, species will be improved.

Reliability of the population estimates

Since there were limitations in the data used to produce the population estimates, a reliability score is provided to give some indication of the level of accuracy that is thought to have been achieved. The scores are as follows:-

1. A widely distributed species for which there had been a recent census or for which there were regular counts that were believed to have a high degree of accuracy, or a rare species for which most of the populations were thought to be known and regularly monitored. For these species it was believed that any improvements in census techniques were unlikely to alter the population estimate by more than 10% either way.
2. A widely distributed species for which the population estimate was based on good data on population densities in different habitat types, or a rare or local species for which the estimate was based on a good understanding of the factors that were limiting its distribution or numbers, and good information on population densities in a reasonable range of habitat types. For these species, there was scope for improving the population estimate, but any improvements were unlikely to be substantial, and any resulting change in the population estimate would probably be less than 25%.
3. A widely distributed species for which the population estimate was based on a limited amount of data on population densities in different habitat types, or for which the population estimate was obtained by scaling abundance relative to a species for which there was a reasonable population estimate. For rare species, the estimate was based on only a limited knowledge of the factors limiting its distribution or numbers, or on only a few density estimates from a limited range of habitats. Any improvement in the population estimate could result in a change of up to 50%, but on current knowledge it was considered unlikely that the estimate would be wrong by a greater margin.
4. An estimate based on a very limited amount of information on the species, for which there was a need for much more information on either its distribution in Britain, or population densities in a variety of habitat types, or relative abundance. Any improvement in knowledge would greatly improve the basis on which the estimate was achieved, but may not necessarily have made a

substantial difference to the estimate presented here, since all species have been ranked in order of relative abundance and its position in the rankings was thought to be correct.

5. A species for which there was so little information on its distribution and/or abundance in different habitat types, and for which the data were so inadequate or biased, that it was not possible to scale its abundance relative to other species reliably. For these species the estimate was believed on subjective criteria to be within the right order of magnitude, but no greater degree of accuracy was thought to have been achieved.

Of the 65 species and sub species covered in the review, for one there was no population estimate, for ten the estimate was graded 1, for ten it was graded 2, for 20 it was graded 3, for 19 it was graded 4, and for five it was graded 5.

Population trends and factors likely to affect population size

Population size *per se* is not the only, nor necessarily the major, factor to consider when assessing the status of British mammals, and population trends and factors likely to affect future population size are at least as important. Thus absolute numbers are often considered to be less important than the degree of habitat, and hence population, fragmentation (see review by Bright 1993). So for each species the available information on population trends, and significant factors that might affect future population size either locally or nationally, are reviewed. For the latter, only factors likely to have an impact on population size, rather than normal mortality factors, were considered. Since there are so few data on population size or trends for many species, any assessment of the factors likely to have an impact on population size are, of necessity, based on subjective, rather than objective, criteria. Hence the sections on population threats for each species must be viewed as a preliminary assessment of perceived, rather than known, risk. Methods of assessing change in population size, and the value of conservation measures to improve the status of threatened species of mammal, are discussed by Jefferies & Mitchell-Jones (1993).

Species accounts

Order: Marsupialia

Red-necked wallaby *Macropus rufogriseus*

Status: Introduced; two recent populations in England were founded in 1940 and a third in about 1985, and one in Scotland in 1975.

Distribution: Colonies are still present in the Peak District and on the island of Inchconachan, Loch Lomond, Scotland; the colony in Ashdown Forest, Sussex, and that near Teignmouth, Devon, are now extinct. In addition there are occasional escapes from wildlife collections.

Population data: The last record from the Sussex colony was in 1972 and this population is assumed to be extinct. The Teignmouth population originated from an unknown number of animals that escaped, probably early in 1985; an animal killed on the road in 1994 was believed to be the last survivor (C.J. Wilson pers. comm.). Survey techniques for the Peak District population are detailed in Yalden (1988). The population was divided into two, and each sub-population counted on a separate day. The total population size is based on 'best counts'. Intensive field work in 1978 showed that, with enough visits, 'best counts' seemed to provide a total population estimate, and no other individuals were known to occur away from the two sub-populations. In January 1993 the Peak District colony stood at only three animals

Brown long-eared bats were not known from Orkney and Shetland (Millais 1904-1906), but were recorded from the north of mainland Scotland, where they outnumbered all other species, and were recorded from the Isle of Arran, Islay, Mull and North Uist (Barrett-Hamilton & Hinton 1910-1921). During this century brown long-eared bats have undergone a long-term decline in relative abundance, and also probably in their distribution. The reasons for this decline are unknown.

Population trends: These are currently unclear. Brown long-eared bats are very dependent on roof spaces and so, during the last 30 years, they have been at risk from timber treatment in buildings using organochlorine pesticides. Since this was coupled with the loss of deciduous woodland, a habitat of great importance to brown long-eared bats, R.E. Stebbings (pers. comm.) argues that the species has declined in recent decades and that this decline may have been substantial.

Grey long-eared bat *Plecotus austriacus*

Status: Native; very rare, and only a few small colonies are known.

Distribution: Found only in Devon, Dorset, Hampshire (including the Isle of Wight) and Somerset.

Population data: These are all subjective, based on the limited distribution and paucity of records. Speakman (1991), A.M. Hutson (pers. comm.) and A.J. Mitchell-Jones (pers. comm.) all suggested that the population is around 1000 individuals, and R.E. Stebbings (pers. comm.) suggested that the figure was in the range 500-1500.

Population estimates: A total pre-breeding population of about 1000, all in England. In view of the limited range and information on the species, this is the best available estimate.
Reliability of population estimate: 3.

Historical changes: Unknown, but in view of its largely Mediterranean distribution, it has probably always been rare in Britain.

Population trends: Any long-term population trend is unknown. Grey long-eared bats are generally rare in north-west Europe, but common in southern areas, particularly around the Mediterranean. Thus they are vulnerable to harsh winters, and in the cold winter of 1962/1963 one colony in Dorset declined from 22 to 4 individuals (Stebbings & Griffith 1986). Three colonies in Dorset and one in north-west Devon have all declined to extinction in the last 20-30 years (R.E. Stebbings pers. comm.).

Order: Lagomorpha

Rabbit *Oryctolagus cuniculus*

Status: Introduced; in most areas numbers are expanding again following a dramatic decline after the introduction of myxomatosis.

Distribution: Rabbits are widespread throughout mainland Britain up to the treeline and on most small islands but absent from Rum and Tiree. Their distribution on islands is documented by Flux & Fullagar (1992).

Population data: The most suitable habitats are areas of short grasses, whether these are natural or agricultural. However, they are found in a wide variety of habitat types. Densities vary seasonally, with numbers relatively stable over winter (<1 to 15 per ha) followed by highly variable summer peaks (<1 to 40 per ha) (Tittensor 1981). Summer peaks are higher

on sandy soils, and over-winter numbers are higher on sand and chalk than on clay soils (Cowan 1991). The following density estimates were available for over-wintering rabbit populations in specific habitat types: 8·4 per ha on chalk grassland, Oxfordshire, based on a capture/mark/recapture study averaged over six winters (Cowan 1984); 14-22 per ha on sand dunes, Holy Island, Northumberland, based on marking and population counts (MacDonald 1989); 2·1 per ha on sand dunes, East Lothian, using Leslie's trap out method (Kolb 1991a); 1·7 per ha on forestry/hill grazing, Borders, using marking and Leslie's trap out method (Kolb 1991b); hill farm/open grazing, Borders, using Leslie's trap out method - (H.H. Kolb pers. comm.); 12·6 per ha on grassland/broom, Strathmore, Tayside, using Petersen/Bailey capture/mark/recapture calculations (H.H. Kolb pers. comm.).

These figures are for densities that are locally quite high, and if they were typical for all the areas of comparable habitat, they would suggest an over-winter rabbit population approaching 100,000,000. This would be comparable to the estimates of the rabbit population before the introduction of myxomatosis. Whilst the rabbit population is rapidly recovering, it has not reached pre-myxomatosis levels (see below). Thus, for the population estimate, the following density figures were used: 5 per ha for scrub, bracken, sand dunes and sloping coastal cliffs; 2·5 per ha for parkland, lowland heaths, lowland grasslands and arable land; 2 per ha in semi-natural broadleaved, semi-natural coniferous, semi-natural mixed and recently felled woodlands, and broadleaved, coniferous, mixed and young plantations; 0·5 per ha in upland unimproved grassland; and 0·1 per ha in heather moorlands.

Population estimates: A total pre-breeding population of about 37,500,000; 24,500,000 in England, 9,500,000 in Scotland and 3,500,000 in Wales. Compared to pre-myxomatosis days, there has been a very significant change in the relative abundance of rabbits in each of the three countries, with the change being most dramatic in Wales (see below). **Reliability of population estimate:** 3.

Historical changes: Rabbits were originally held in warrens, and there were only substantial increases in wild populations from the mid-18th century onwards, when changes in agricultural practice created favourable habitats and an increased interest in game led to intensive predator control (Thompson & Worden 1956). In Scotland, rabbits were for a long time mainly confined to a few islands and to coastal sand dunes. Early in the 19th century rabbits were rare north of the Tay and Clyde, and their range extension throughout Scotland was gradual (Millais 1904-1906). The natural spread of rabbit populations can be quite slow (Kolb 1994), and their spread in Scotland was largely due to 18th and 19th century introductions throughout the highlands and the north-west (Lever 1977). A similar pattern of events occurred in Wales. Until the early 19th century rabbits were mostly confined to large warrens on islands, but introductions to the mainland, coupled with agricultural changes and large-scale predator control, led to rapid increases in numbers (Lever 1977). At the turn of the century rabbit numbers were high, and average bag returns remained high until the onset of myxomatosis in 1953 (Tapper 1992). At that time the British rabbit population was estimated to be 60,000,000-100,000,000, with about 40,000,000 being culled each year for the meat and fur trade (Thompson & Worden 1956). Thompson & Worden (1956) believed that the upper limit was a conservative estimate, and that the spring and summer population was in excess of 100,000,000 rabbits. Myxomatosis destroyed over 99% of the British rabbit population (Thompson 1956). Assuming that the original estimate was reasonably accurate, this implies that fewer than 1,000,000 rabbits were left in Britain following the introduction of myxomatosis.

Population trends: For an *r*-selected species such as the rabbit, it is difficult to be precise about trends, because even several years of decline can be recouped remarkably quickly. However, its powers of recruitment are not as high as sometimes suggested by calculations based on counts of corpora lutea or of late term foetuses, since these ignore the rate of nestling mortality, which can be considerable (Bell & Webb 1991).

Since the advent of myxomatosis, bag records suggest that nationally the rabbit population has steadily recovered (Tapper 1992), although locally this may not be the case. On Forvie

sand dunes, Grampian, rabbit numbers have declined from a peak in 1976, with considerable and possibly regular fluctuations in abundance (I.J. Patterson pers. comm.). Long-term data from agricultural areas show that in 1953, before the onset of myxomatosis, 94% of farm holdings in England and Wales had rabbits on cultivable land. By 1970 this figure had only recovered to 59% (Lloyd 1970), and by 1986 rabbit numbers were still only around 20% of pre-myxomatosis levels (Cowan 1991). Assuming that the estimate of up to 100,000,000 rabbits pre-myxomatosis was correct, Cowan's (1991) estimate suggests that the rabbit population in the mid-1980s was around 20,000,000.

It is also clear that the rabbit population is still well below that recorded before myxomatosis. The National Game Bag Census data (Tapper 1992) suggest that rabbit numbers are currently between a third and a half of the pre-myxomatosis numbers, which is supported by the estimate presented here. In Scotland, 92·9% of farms had rabbits pre-1954. The corresponding figure was only 60·7% in 1970 and 80·7% in 1991. The number of farms with serious infestations fell from 54·9% pre-1954 to 0·4% in 1970, and recovered to 16·6% in 1991 (Kolb 1994). In addition to dramatic changes in rabbit numbers, there has also been a significant change in the distribution of rabbits. During the 1970s eastern and south-eastern regions consistently showed the highest rises in rabbit numbers, the south-west intermediate, and Wales and the north relatively low increases (Trout, Tapper & Harradine 1986; Kolb 1994). Pre-myxomatosis populations were high in the south-west and in Wales (Thompson & Worden 1956), whereas Wales now has one of the lowest rabbit populations. In Scotland, there have consistently been more farms with rabbit infestations in the east, Highlands and north-east. These areas also had more farms with serious infestations when compared with the rest of the country, both before and after myxomatosis (Kolb 1994). Trout *et al.* (1986) concluded that in many areas the carrying capacity for rabbits had still to be reached, and so further population increases are to be expected. All these observations agree well with the estimates for the size and distribution of the rabbit population presented here.

Population threats: Fresh outbreaks of myxomatosis cause local depletions, and rabbit control measures can suppress populations locally. Whilst myxomatosis is still prevalent in many areas, mortality during such out-breaks has fallen to 40-60% (compared with over 99% at the start of the epidemic) as a result of increasing levels of genetic resistance (Ross 1982; Ross & Sanders 1984). Also, from 1992 there have been reports of viral haemorrhagic disease in wild rabbits in Britain (Duff *et al*. 1994); whether this calicivirus will have a significant impact on British rabbit populations remains to be seen.

Annual changes in reproductive success, and particularly the onset of the breeding season, are affected by climatic factors, especially higher minimum ground temperatures and hours of daily sunshine in winter. Following mild winters, the first young of the year appear above ground about two months earlier (in February rather than April), and one study found that, over four years, annual average reproductive success per adult female ranged from 6·1 to 10·1 (Bell & Webb 1991). These data demonstrate the possible effects of predicted global warming on the productivity of rabbits, and hence the potential for an increase in rabbit numbers.

Brown hare *Lepus europaeus*

Status: Probably introduced to Britain by the Romans. There have been many subsequent introductions to islands. Common but has undergone a substantial decline this century.

Distribution: Widespread, and most common in agricultural areas. In Scotland brown hares are absent from the north-west and western Highlands, where they are replaced by mountain hares on heather moorland. Present on Anglesey and the Isle of Wight; they have been widely introduced to Scottish islands (Millais 1904-1906; Barrett-Hamilton & Hinton 1910-1921).

Population data: Three population estimates are available. The first was obtained by Tapper & Stoate (1992). Hares were counted on 12 areas between January and March 1988 and

1989, using spotlight counts as described by Barnes & Tapper (1985). These gave an absolute estimate of brown hare density within a given area; for each area the cropping pattern and presence or absence of a gamekeeper were recorded. From these an index of field size and habitat diversity was calculated. Habitat data from the Institute of Terrestrial Ecology's land classes were then used to calculate a relationship between an index of crop diversity and brown hare numbers, which in turn was used to estimate the number of brown hares that would be present both with and without the presence of gamekeepers to control the numbers of predators. By this means, Tapper & Stoate (1992) estimated that the number of brown hares in Britain in late winter lay between 1,250,000 and 1,911,000, the figures being, respectively, in the absence or presence of predator control. Whilst this approach gave a reasonable estimate of total population size using the data then available, it had its limitations. First, only a minority of land classes were surveyed for brown hares, and the results from 12 sample sites predominantly in southern and eastern England were extrapolated to produce a national estimate of hare numbers. Also, the 12 survey sites chosen were areas with known hare populations, and generally areas with reasonable population densities; no areas with few or no hares were included in their calculation of the relationship between hare numbers and habitat features. This is likely to produce an over-estimate. A second estimate was derived by the Game Conservancy from the National Game Bag Census, extrapolated to the whole country, based on the assumption that about 40% of hares are shot during winter hare shoots (Tapper & Stoate 1992). This suggested a population of about 1,000,000 hares before the main culling season.

A third estimate was produced by M. Hutchings & S. Harris (unpubl.), who organised a stratified national survey in which 734 randomly selected 1 x 1 km squares were surveyed for hares. This survey was undertaken in the winters of 1991/1992 and 1992/1993. A transect in each square (mean length 2950 m) was walked three times between mid-October and mid-January, and the position of each hare seen was recorded in relation to the transect line. The data were analysed using the program DISTANCE, an updated version of Burnham, Anderson & Laake's (1980) program TRANSECT. This program was used to calculate population densities in each of the four land class groups, based on a detection function. By this means, a mid-winter brown hare population of 817,500 $\pm$ 137,250 (95% confidence limits) was obtained. This figure is lower than the other two because a much wider range of habitats were sampled, including many 1 x 1 km squares with few or no hares. The relatively large 95% confidence limits are due to the very clumped distribution of the hare population, so that comparable habitats can have widely different hare numbers. This clumping is due to various anthropogenic factors, most notably the willingness of the land owner to tolerate the presence of hares. In many areas hare numbers are drastically reduced so that poachers are not attracted onto the land (M. Hutchings & S. Harris unpubl.).

Population estimates: A mid-winter population, at the start of the breeding season but before the onset of the main hare-culling season, of about 817,500; 572,250 in England, 187,250 in Scotland and 58,000 in Wales. Organised shoots at the end of the winter may lead to a 40% decline (Tapper & Stoate 1992). **Reliability of population estimate:** 2.

Historical changes: At the turn of the century brown hares were considered to be abundant throughout England, Scotland and Wales except on the higher parts of mountains, and in the 19th century were even found on open areas in London (Barrett-Hamilton & Hinton 1910-1921). Reports in the *Victoria Histories of the Counties of England* show that they were certainly much more abundant in the western parts of England than at present, but at the start of this century declines were already beginning in the south-west (M. Hutchings & S. Harris unpubl.). Prior to the Ground Game Act of 1880, the abundance of hares in some districts was described as 'quite extraordinary', but the Ground Game Act removed the protection that was enjoyed by brown hares and is thought to have led to a dramatic decline in numbers, followed perhaps by an increase around the turn of the century. Thorburn (1920) described brown hares as plentiful in cultivated areas, especially grasslands, when not driven away by persecution.

More recently, National Game Bag Census data (Tapper 1992) have shown that hare bags were highest in the early part of the century, and that after the late 1920s they declined until the latter half of the Second World War. Thereafter numbers increased until 1960 (but not to pre-1920 levels). This increase in the later half of the 1950s may have been aided by the decline in rabbit numbers, providing a niche into which the hares expanded (Rothschild 1963). This was followed by a further decline in hare numbers during the 1970s and 1980s. This decline occurred in virtually all regions (Tapper 1992). Tapper & Parsons (1984) concluded that it represented a real decrease in abundance, and that the hare bag was declining at a more or less steady rate when considered nationally. However, autumn counts from 1976-1992 in the Dane Valley, Cheshire, showed no evidence of any overall population change, although there were marked inter-annual variations (D.W. Yalden unpubl.).

Population trends: Many estates no longer hold as many large hare shoots as previously, which complicates the interpretation of the National Game Bag Census data. However, the available information suggests that overall hare numbers have remained constant for the last ten years (mean 2·80 shot per km^2 for the years 1983-1987, mean 2·95 per km^2 for the years 1988-1992), although there will be significant annual and seasonal fluctuations due to the effects of summer weather conditions on breeding success. Hunting records also suggest that, nationally, hare numbers have remained constant over the last few years (Stoate 1993). This index is based on the number of hares seen whilst out hunting with packs of beagles and is the average of all hunting trips for the entire season, and then averaged between packs. Thus the index is based on many data which are independent of the game returns. Neither source of data shows any change in hare numbers over the last ten years. This general pattern of population change over the last thirty years has occurred throughout much of Europe, and generally hare numbers appear to have stabilised at a relatively low level (S.C. Tapper pers. comm.). However, a recent national survey suggested that population declines may be continuing since hares are heavily culled in many areas in eastern England because they attract poachers. These declines would not be detected by the National Game Bag Census and hunting data, which are biased towards large keepered estates and hunting areas where hare populations receive a degree of protection. Also, it appears that hares may be continuing to decline in low density areas in the west, and again the National Game Bag Census and hunting data are not suitable for monitoring changes in such low density hare populations (M. Hutchings & S. Harris unpubl.).

Population threats: Numerous explanations have been advanced for the decline in brown hare numbers, but none is in itself totally satisfactory. Barnes & Tapper (1986) concluded that there was a boom in hare numbers in the late 1950s and early 1960s as a consequence of the myxomatosis epizootic in rabbits, which removed a potential competitor. However, the subsequent decline in hares could not be explained adequately either by the increase in rabbits or a series of poor breeding seasons for the hares. High hare numbers in bag returns are associated with mild springs (Barnes & Tapper 1986), and mild autumns appear to lengthen the breeding season, sometimes significantly (Hewson & Taylor 1975). Thus weather can have a dramatic impact on hare breeding success.

The most generally accepted cause of the decline of the brown hare is change in the agricultural ecosystem (Tapper & Barnes 1986; Tapper & Stoate 1994). This in itself does not explain the actual cause of the decline. We do not know the relative importance of habitat simplification, use of agrichemicals, changes in farming practice such as cutting silage instead of hay, increasing use of complex machinery, etc. All these factors are likely to have played a role in the decline of hares, and their relative impact will vary in different regions of the country. In addition, hare numbers have increased where fox *Vulpes vulpes* numbers have been reduced experimentally (Tapper, Potts & Brockless 1991), but whether predation has played a role in their general decline is currently unclear, as is whether hares are now more vulnerable to predation as a consequence of habitat simplification (Harris & Saunders 1993). Whilst the reasons for the decline in brown hare numbers are probably complex, and at present are poorly understood, it is clear that many factors are involved. Anecdotal observations suggest that the five-year set-aside scheme led to an increase in brown hare numbers in some areas.

Mass mortalities of brown hares are sometimes reported from parts of East Anglia, most often in the summer. A number of factors may be responsible: food shortage due to agricultural changes, such as the change from growing spring to winter cereals (G. McLaren & S. Harris unpubl.), grass sickness (Whitwell 1991; Griffiths & Whitwell 1993), or European brown hare syndrome (Duff *et al*. 1994), although this disease is most prevalent in the autumn (Gavier-Widén & Mörner 1991), whereas the mass mortalities occur earlier in the year. Their impact on brown hare populations is at present unknown.

In the Royal Museum of Scotland, Edinburgh, there are about 20 skins and skulls of specimens that appear to be brown hare/mountain hare hybrids, although whether they actually are hybrids has yet to be determined (Balharry *et al*. 1994). These came from Argyll, Ayrshire, Dumfriesshire and Peeblesshire, i.e. those parts of Scotland where mountain hares were introduced in the middle of last century. Whether hybridisation is frequent in these areas needs to be determined (A.C. Kitchener pers. comm.). However, the current evidence suggests that wild hybrids occur only rarely, and so do not represent any real threat to the population (Balharry *et al*. 1994).

Mountain hare *Lepus timidus*

Status: Native but widely introduced outside its natural range; locally common in some upland areas.

Distribution: In the British Isles, mountain hares are indigenous only in the Highlands of Scotland; all the populations south of the Clyde and Forth are the result of introductions. Mountain hares are most numerous on grouse moors in north-east Scotland, and uncommon in west Scotland. They were present on Orkney in mediaeval times and died out; the modern population is the result of a recent introduction. Other introductions, mostly during the 19th century, were to Eigg (now extinct), Hoy, Islay (now extinct), Jura, Mull, Outer Hebrides, Raasay, Scalpay, Shetland and Skye. They were introduced into Ayrshire, Lanarkshire and Peeblesshire in the 1830s and 1840s, from where they dispersed widely (Barrett-Hamilton & Hinton 1910-1921), and to the Pennine area of South Yorkshire and Derbyshire about 1880. Introductions to the Cheviots and Lake District have not persisted. The population in North Wales, originating from introductions to the Vaynol estate near Bangor about 1885, is now extinct (Hewson 1991). No mountain hares have been reported in Wales for at least 15 years (R. Lovegrove pers. comm.).

Population data: Mountain hare populations are very localised; they reach the highest densities in north-east Scotland, and are particularly scarce in north and west Scotland (Watson & Hewson 1963). Population densities range from 3 to 46 per km^2, depending upon habitat type. The highest densities occur on heather moors overlying base-rich rocks, with the lowest densities where there are acidic rocks; locally densities may reach 300 per km^2 (Hewson 1991). In Scotland there are approximately 12,000 km^2 of heather moorland. Using a simple population estimate based on this area of heather moorland, and a mean density of 30 hares per km^2, the population in Scotland would be about 300,000. An alternative estimate was based on the following assumptions. Arnold (1993) recorded mountain hares in 363 10 x 10 km squares, including the Scottish islands and the Pennines. It was assumed that they were found in these squares but nowhere else, that they occurred at an average density of 2 per km^2 on the Scottish islands and north-west of the Great Glen (an area of about 5500 km^2). Over the rest of Scotland a density of 20 per km^2 was assumed where they are present (an area of about 35,200 km^2). However, hares are likely to be absent from large areas of this range which do not consist of heather moorland. Based on these two mean densities, and assuming that only half the range was suitable for hares, the population in Scotland would be about 360,000. In the Peak District, extensive surveys showed that the annual late winter density varied between 1·4 and 3·3 per km^2 (average 2·1 per km^2) (Yalden 1984a). Since they occupy an area of 246 km^2, the late winter population for the Pennines is about 500 animals.

Population estimates: A total pre-breeding population of about 350,000 animals, split into several sub-populations; 500 in England, 350,000 in Scotland and none in Wales. **Reliability of population estimate:** 3.

Historical changes: Millais (1904-1906) described the distribution of mountain hares at the turn of the century. The National Game Bag Census records (Tapper 1992) show that high numbers were shot around the turn of the century, and even more after the First World War. They reached a peak around the early 1930s. Low numbers during the Second World War were followed by a recovery by 1950 to some 50% of those in the 1930s. There was then a decline which lasted from the mid-1970s to the early 1980s, since when the numbers shot have increased to equal the bag levels of the 1950s. At present the reasons for these population changes are unknown.

Population trends: Tapper (1987) showed that, whilst just over half of the populations he examined show irregular fluctuations in numbers, the rest showed a weak tendency towards regular population cycles with a periodicity of about 9·5 years. With species such as the mountain hare that fluctuate widely in number, it is difficult to detect long-term population changes.

Population threats: The main threat is probably population fragmentation. Populations in the Pennines, Mull, Orkney and Shetland are all relatively small and isolated. Furthermore, most recoveries of marked mountain hares are close to the point of capture; the maximum recovery distance in one study was 12 km (Hewson 1990a). Assuming that mountain hares will rarely cross more than 20 km of unsuitable habitat, the Scottish population is effectively fragmented into a number of sub-populations, particularly in the north and west (see map in Arnold 1993), and some of these could be quite small. The number of introduced populations that have died out shows the vulnerability of small mountain hare populations - see Millais (1904-1906). The Pennine population is also potentially vulnerable because of its small size, wide distribution, and the pressures placed upon it by high visitor use of the area.

It is possible that mountain hares are susceptible to climatic instability, and that adverse weather conditions affect juvenile mortality. Thus there is considerable variation in recruitment rate from place to place and possibly also from year to year (Flux 1970). Such chance events will enhance the vulnerability of small populations. Also, reduced management of heather moorland, and substantial reductions in the area of heather resulting from increased sheep-grazing, pose a threat to mountain hares (Anderson & Yalden 1981; Hewson 1984a). However, current Common Agricultural Policy reforms may reduce grazing pressure in the uplands, as may some of the new Environmentally Sensitive Areas. The new Southern Uplands Environmentally Sensitive Area is specifically targeted at increasing the regeneration of heather moorland, and may thus benefit mountain hares (J. & R. Green pers. comm.).

In the Royal Museum of Scotland, Edinburgh, there are about 20 skins and skulls of specimens that appear to be brown hare/mountain hare hybrids. These come from Argyll, Ayrshire, Dumfriesshire and Peeblesshire, in those areas where the mountain hare population was established following introductions to Ayrshire in the middle of the last century (Hewson 1991). Whether hybridisation is frequent in these areas needs to be determined (A.C. Kitchener pers. comm.). However, the current evidence suggests that wild hybrids occur only rarely, and so they do not represent any real threat to the population (Balharry *et al*. 1994). Moreover, Irish hares were introduced to south-west Scotland in about 1923 (Hewson 1991), and J. & R. Green (pers. comm.) have suggested that the colour varieties seen in the hares in that part of Ayrshire may be the descendants of that introduction. Certainly, Irish hares have been introduced to a number of areas where mountain hares were present, e.g. Mull and Vaynol Park, and the two races remained distinct. The colony on Mull, for instance, was introduced around 1860, and the descendants were still identifiable fifty years later (Barrett-Hamilton & Hinton 1910-1921). Thus it is possible that Irish hares are still recognisable in Ayrshire 70 years after their introduction. These observations suggest that the two races may be reproductively isolated (Balharry *et al*. 1994).

introduced by the Vikings in the 9th and 10th centuries AD (Campbell 1974). When St Kilda was evacuated in 1930, the Soay sheep were left on the island of Soay, as were the flock of primitive blackface sheep on Boreray. In 1932, 107 Soay sheep were transferred from Soay to the larger island of Hirta by the St Kildans, who returned annually to tend the sheep on Boreray and Hirta until the outbreak of the Second World War, when the Soay sheep on Hirta were said to number about 500. In 1947 the flock was said to number 400-450 and in 1948 650-700 (Lever 1977). Annual counts from 1955 to 1973 showed that the Soay population on Hirta fluctuated between 610 and 1783 (Boyd 1974). Nine counts of the Boreray blackface sheep between 1951 and 1971 showed that the minimum flock size varied between 330 and 466 (Lever 1977). In 1934 six Soay sheep were introduced to Skokholm, in 1944 eight were introduced to Cardigan Island, two to Middleholm Island in 1945, four to St Margaret's Island near Tenby in 1952, and in 1958 a few were introduced to Skomer. In 1975 the only Welsh island still to have Soays was Cardigan, where there were 80. Soay sheep were introduced to Lundy around 1927, and by 1959 numbered over 80. The Soays on Ailsa Craig were introduced direct from St Kilda in the 1930s; in 1956 they numbered 14 (Lever 1977), but do not persist today.

Population trends: The Soay sheep on Hirta, and possibly those on Soay, show cyclical changes in numbers and these are probably density-dependent. Thus on Hirta, high winter mortality occurs every three to four years following summers when population density exceeds 2·2 sheep per ha. During these die-offs, more than 50% of adults, 70% of yearlings and 90% of lambs die and population density falls by around 65% (Clutton-Brock *et al*. 1992). Despite these perturbations, there are no long-term population changes. On Hirta, changes in population size occur as a result of high over-winter mortality from starvation and this is particularly pronounced among lambs and rams (Clutton-Brock *et al*. 1991), thereby giving an adult population with a varying bias towards females. Recent calculations, taking into account this unusual demography, suggest that the effective population size of the Hirta Soay sheep is in the range 200-250 (D.R. Bancroft pers. comm.).

Population threats: None known. Despite their isolation and population dynamics, the Soay sheep on Hirta have substantial genetic variation at the phenotypic and molecular level (Jewell, Milner & Boyd 1974; J.M. Pemberton & D.R. Bancroft pers. comm.), a conundrum that is the subject of current research. There is the potential threat from diseases or parasites introduced from the mainland. An example is the presence of the nematode *Nematodirus battus* on the island. It was first identified in England in 1951, and it is unclear whether the parasite was a recent introduction or had previously been over-looked. Yet this parasitic worm was found in most of the 120 Soay sheep carcasses examined on Hirta in 1989/1990, despite these sheep being separated from mainland stocks since the 1930s (Gulland 1991; I.R. Stevenson pers. comm.). Also, Gulland (1991) reported that 50% of the Soay sheep examined had cysts of the tapeworm *Taenia hydatigena*, yet there are no carnivores on the island and the last resident dog left with the islanders in the 1930s.

Discussion

Current status

It should be clear from the extent of extrapolation required to achieve most of our population estimates that we do not intend the figures to be accepted uncritically. However, we feel that they indicate the likely order of magnitude of the numbers of each species, and therefore they have allowed the species to be ranked in order of abundance much more objectively than has been done previously (Table 14). One problem with this ranking, however, is that species should not be compared on a simple numerical scale, especially where a relatively non-mobile species is isolated into a number of sub-populations. Large species obviously have a greater biomass and economic impact, and so whilst roe deer and common dormice may be of comparable numerical status, they are very different in both biomass and economic importance. Equally, they differ in mobility, and so in their vulnerability to habitat fragmentation. This problem was considered by Bright (1993), who used five life-history

traits to identify the species of mammal most at risk from habitat fragmentation. These were primarily low density, slow breeding, often poorly mobile species associated with semi-natural habitats, and included all riparian mammals.

For our assessment of the pre-breeding population of each species, a reliability grading has been included, and the implications of these gradings need to be considered in relation to the ranking in Table 14. For example, the field vole is listed as the commonest British mammal, but on the basis of one of the least reliable population estimates because population size had to be calculated relative to other species of small mammal. In general, the rarer species are often those with the most reliable population estimates, and confidence can be placed in the order of magnitude of these estimates. The greatest problems arose with the commonest and most widespread species. Invariably, population sizes were calculated from only a few density estimates, and in all but a few cases these did not include more than a few habitats and rarely included any from Scotland or Wales. Other species for which the estimates have a low reliability rating are those which are relatively thinly spread over a wide variety of habitats, such as the hedgehog. Other problems occurred for species which locally can be very abundant, yet for which typical densities were virtually unknown, such as the harvest mouse.

Thus one aspect of this exercise has been to highlight the lack of basic data on population densities for many, if not most, species of mammal in Britain. More field data, particularly from unusual habitats and/or from Scotland and Wales, will allow the estimates for all species to be improved, and increase the confidence that can be placed both on individual estimates and the ranking of all the species.

Changing status

In judging conservation priorities, absolute population size is one important criterion, but its use is tempered by what is known of recent changes in status. No species has a stable population size, and we have tried to document the likely pattern of recent trends. Many species were reduced by persecution (e.g. polecat) or overhunting (e.g. roe deer) in previous centuries, and are slowly recovering their range and status. Others have been more recently reduced by pollution (e.g. otter) or disease (e.g. rabbit). What time span is important in assessing change? We have taken the last 30 years, that being the period over which the Biological Records Centre has been accumulating data (Arnold 1993). This at least provides some possibility of measuring trends, and this period is also well-covered by the Game Conservancy Trust's National Game Bag Census (Tapper 1992). This choice of time frame produces some surprises. The rarer carnivores, including the otter, have probably been increasing their ranges during this period, albeit from low levels; the greater horseshoe bat has probably been stable; and the species of most conservation concern in respect of recent declines are the red squirrel and the water vole.

Relative status

With the mammals arranged in relative order of abundance, it is possible to make comparisons with other vertebrates, particularly those of conservation interest. For birds, Schedule 1 of the Wildlife and Countryside Act 1981 lists those of special concern, and they are discussed further by Batten *et al*. (1990). Mostly, they have fewer than 1000 breeding pairs. For example there are 52 pairs of red kite *Milvus milvus*, 75 'pairs' of marsh harrier *Circus aeruginosus*, and 424 pairs of golden eagle *Aquila chrysaetos*. Few British mammals are so rare, but also few, if any, are so well documented, and none are so mobile. Generally, the rarest mammals are an order of magnitude more numerous. At the other end of the range, the most abundant breeding birds are the wren *Troglodytes troglodytes* with 7,100,000 pairs, the chaffinch *Fringilla coelebs* with 5,400,000 pairs, the blackbird *Turdus merula* with 4,400,000 pairs and the robin *Erithacus rubecula* with 4,200,000 pairs (Gibbons, Reid & Chapman 1993). Similarly, the commonest mammals, (rabbits and five of the small mammals) are roughly an order of magnitude more common. Of comparable abundance, however, are some of the domestic mammals. There are 41,050,000 sheep *Ovis aries*,

10,330,000 cattle *Bos taurus* (Government Statistical Service 1992), 7,300,000 domestic dogs *Canis familiaris* (Pet Food Manufacturers' Association pers. comm.) and 7,600,000 domestic cats *Felis catus* (Cats Protection League 1993). The abundance of these species must reduce their wild counterparts by competitive exclusion, e.g. dogs and urban foxes (Harris 1981) or sheep and red deer (Clutton-Brock & Albon 1989). The relative ecological impacts of wild and domestic mammals may also be judged from these population sizes. Of the 41,050,000 sheep in Britain, approximately 60% are in upland areas. These clearly have more impact on upland ecosystems than around 360,000 red deer.

Three species of reptile and amphibian are rare enough to be of conservation concern. It has been estimated that there are 20,000 natterjack toads *Bufo calamita*, 7000-8000 sand lizards *Lacerta agilis* and 2000 smooth snakes *Coronella austriaca* in Britain (Nature Conservancy Council 1983). These compare numerically with the rarer mammals, but suffer even more the problems imposed by small size and limited mobility.

European status

The conservation status of each species under British and European legislation is shown in Table 15. Whilst this summarises the protection thought to be required for each species, for most the information on their status across Europe is even worse than for Britain, which precludes most direct comparisons of population size. However, we can put British species into a European perspective by assessing the proportion of their range which Great Britain represents, since detailed distributions were given by Niethammer & Krapp (1978-1990) for the Insectivora, Rodentia and Artiodactyla, and all species were mapped by Corbet (1978). For this, we have considered 'western Europe' to include that area west of the former USSR, a total area of 4,909,989 km^2 (Novotny & Pankova 1981). Iceland and Ireland have been excluded since, being islands, their mammal fauna is limited. Some northern species extend south to the Alps and Pyrenees, but are largely absent from 'Mediterranean' countries (Albania, Bulgaria, Greece, Italy, Portugal, Spain and what was Yugoslavia), an area of 1,425,082 km^2. Similarly, some southern species are largely absent from 'Scandinavia' (Finland, Norway and Sweden), an area of 1,110,916 km^2. What we will term 'north-west Europe' ('western Europe' minus the 'Mediterranean' and 'Scandinavian' countries) covers an area of 2,373,991 km^2. The area covered in this review thus forms approximately 9·7% of 'north-west Europe', 6·6% of 'northern Europe' ('north-west Europe' and 'Scandinavia'), 6·1% of 'southern Europe' ('north-west' and 'Mediterranean' Europe) and 4·7% of 'western Europe'. These percentages form the basis of the entries in Table 14. A species with more than 4·7% of its European range in Britain is more important from a European perspective than would be expected from the area available. Inherent inaccuracies in these figures (for example, whether the species' range includes islands such as Corsica, Ireland, Sardinia and Sicily) are relatively trivial, given the approximations being attempted.

The insectivores are probably about as common in the rest of Europe as in Britain, though the hedgehog is replaced in the east by the eastern hedgehog *Erinaceus concolor*, and the common shrew is similarly replaced in France, Italy and Spain by three species: Millet's shrew *Sorex coronatus,* the Appenine shrew *Sorex samniticus* and the Spanish shrew *Sorex granarius* respectively. In Europe generally, as in Britain, the water shrew is much rarer than either the common or pygmy shrews. The lesser white-toothed shrew is obviously represented here in a minuscule segment of its overall range, but its density (in the absence of any competitors) is probably much higher in the Isles of Scilly than elsewhere.

Most of the rare Chiroptera are at the edge of their ranges in Britain, and are relatively much more common in southern Europe. This applies particularly to the greater and lesser horseshoe and grey long-eared bats. In a European context, therefore, their British populations are even less significant than suggested by the small proportions of their European ranges which Britain contributes. Bechstein's bat is an exception to this, being rare everywhere. Saint Girons (1973), for instance, listed only 29 records for France, three for Belgium and one for the Netherlands; Benzal, Paz & Gisbert (1991) gave only 17 records for Iberia; and Pucek (1981) gave only 12 records for Poland. The barbastelle bat is as rare in


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State of Nature 2019 **documents how human impacts are driving sweeping changes in wildlife in the UK. The loss of nature affects us all, but the greatest impacts will be upon the lives of young people and generations yet to come, if they have to live in a world impoverished of nature. In recognition of this, we asked some of the UK's most passionate and committed young conservationists to tell us what nature means to them.**

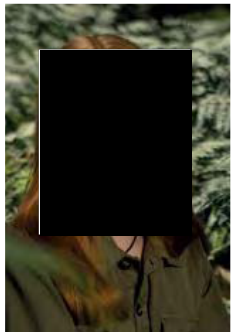
Photo: David J Slater (rspb-images.com)



Swallows

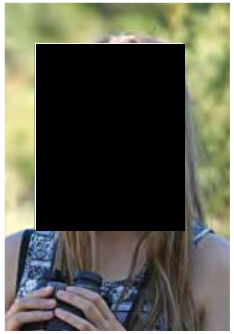
"Nature provides liberation from the hustle of modern society, allowing our attention to shift away from ourselves and focus on the glorious and the sublime. For me it is not a supplement, it is a necessity."

BELLA LACK, 15



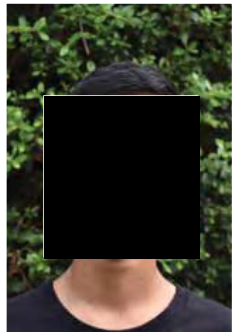
"The revival of the Red Kite, Ospreys bouncing back, breeding Cattle Egrets – many species are bringing new life to the UK but the list of declining species is ever growing. We must unite now to save these species before they're but a distant memory."

DAN ROUSE, 23



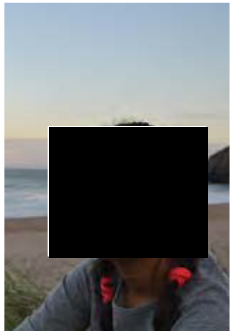
"Thanks to the dedication of schoolchildren and volunteers in the City, wildflower meadows and roof gardens have been created, allowing pollinators, including many butterflies, to thrive in central London. This shows the potential we all have to protect nature, in every place, even in our concrete jungles."

KABIR KAUL,14



"Whenever I have the chance to experience the outdoors, the presence of nature brings a feeling of tranquillity. Seeing the variety of animals and plants that our forests, parks and meadows contain gives me the urge to protect the environment even more."

KHADIJAH HAQ, 14





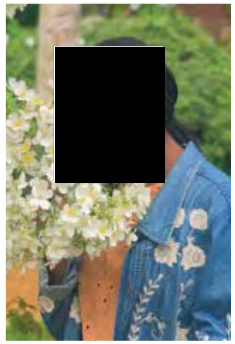
"Nature is our life support system. As an autistic teenager, nature has provided a safe space to which I can crawl into, rejuvenate my spirits and keep me going."

DARA MCANULTY, 15



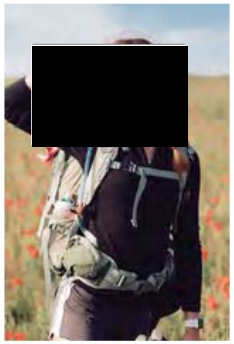
"I wholly believe that it is our duty to protect nature and the environment as a reciprocation to how nature takes care of us every single day. Nature allows us to eat, drink, breathe, live. The least we can do is protect it."

YETUNDE KEHINDE, 17



"My favourite thing about nature is its unmistakable diversity. I am reminded of the woods by the captivating scents of wildflowers; majestic colours of birds soaring high, and simply the soothing symphony of birdsong."

PRINCESS-JOY EMEANUWA, 17



"I worry that people don't recognise how valuable their individual actions are in sparking change. Planting wildflowers, drilling a few holes in bricks, being a bit 'lazier' in the garden, can encourage so much wildlife to your doorstep – it couldn't be easier!"

SOPHIE PAVELLE, 24

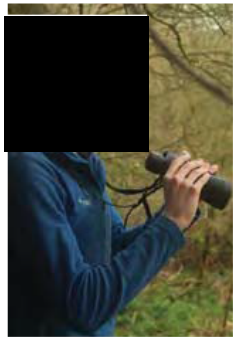
"Nature is important to me because it reminds me to keep going even when things are hard. The fresh air and petrichor give me space to breathe and let go and make me feel better. Nature always gives a solution, and adapts, always trying hard, and I think we all have something to learn from nature."

ESTHER BIRD, 13



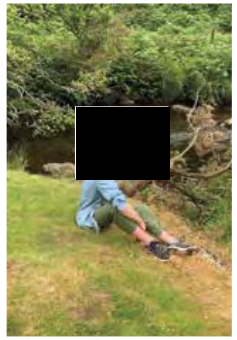
"I have never seen a Hedgehog, although my parents used to see them all the time in the area. Many others my age have had the same experience. I'm worried that we're close to losing them from our countryside forever."

JAMES MILLER, 17



"During my GCSE exams, Dartmoor was a refuge – wading through streams, finding bats in hidden caves and making camps under trees. Immersing ourselves in nature like this is the antidote to our dissociation from the earth that has driven the climate crisis."

SOPHIE SLEEMAN, 17



"It is everyone's responsibility to contribute to help the natural environment. Even the smallest of actions can make the biggest of difference, like rewilding your own garden to make it insect, bird and mammal friendly."

XANDER JOHNSTON, 13



STATE OF NATURE 2019

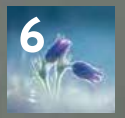
presents an overview of how the country's wildlife is faring, looking back over nearly 50 years of monitoring to see how nature has changed in the UK, its Crown Dependencies and Overseas Territories.

As well as this long-term view, we focus on what has happened in the last decade, and so whether things are getting better or worse for nature.

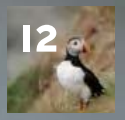
In addition, we have assessed the pressures that are acting on nature, and the responses being made, collectively, to counter these pressures.

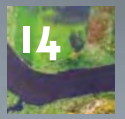


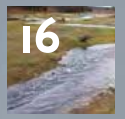
CONTENTS

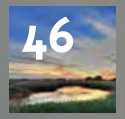

6 Headlines


8 Key findings

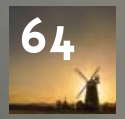

12 Results in more detail

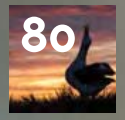

14 Historical change

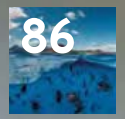

16 Drivers of change

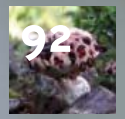

46 Conservation


50 Marine


64 UK countries


80 UK Overseas Territories and Crown Dependencies


86 Essays


92 Appendix

This report has been produced by a partnership of more than 70 organisations involved in the recording, researching and conservation of nature in the UK and its Overseas Territories. These include a broad spectrum of recording societies with expertise on a wide range of taxonomic groups, including our best and least known wildlife; research organisations responsible for gathering and analysing data that advances our knowledge of the UK's nature; conservation charities that take action for all elements of our wildlife and habitats; and the country nature conservation bodies for the UK and it's four countries which have provided evidence and advice in the production of this report.

Unless otherwise stated, all photos are from RSPB Images (rspb-images.com).

This report should be cited as:
Hayhow DB, Eaton MA, Stanbury AJ, Burns F, Kirby WB, Bailey N, Beckmann B, Bedford J, Boersch-Supan PH, Coomber F, Dennis EB, Dolman SJ, Dunn E, Hall J, Harrower C, Hatfield JH, Hawley J, Haysom K, Hughes J, Johns DG, Mathews F, McQuatters-Gollop A, Noble DG, Outhwaite CL, Pearce-Higgins JW, Pescott OL, Powney GD and Symes N (2019) *The State of Nature 2019*. The State of Nature partnership.

Photo: Ben Andrew (rspb-images.com)
Cover photo: Mark Eaton

Much of the content of *State of Nature 2019* can be used to inform on the UK's progress towards the Convention on Biological Diversity's (CBD) Aichi 2020 targets – see pages 90–91 or further details. We have used the CBD's icons for the Aichi targets to indicate when report content is relevant to these targets.



For guidance on how to interpret the results presented in this report, please refer to pages 92–93.

THE HEADLINES

In this report we have collated the best available data on the UK's biodiversity, with a focus on the trends in species as the key evidence of how nature is faring. In addition to assessing the state of nature we have reviewed the pressures acting upon nature, and the conservation response being made to counter these pressures, in order to give a rounded view of the UK's nature in 2019.

Our statistics demonstrate that the abundance and distribution of the UK's species has, on average, declined since 1970 and many metrics suggest this decline has continued in the most recent decade. There has been no let-up in the net loss of nature in the UK.

Prior to 1970, the UK's wildlife had already been depleted by centuries of persecution, pollution, habitat loss and degradation.



Pasqueflower

Photo: Ben Andrew (rspb-images.com)

13%

decline in average species' abundance.

Our indicator of average species' abundance of 696 terrestrial and freshwater species has fallen by 13% since 1970; the rate of decline was steeper in the last 10 years, although not statistically significantly so.

5%

decline in average species' distribution.

Our indicator of average species' distribution, covering 6,654 terrestrial and freshwater species over a broad range of taxonomic groups, has fallen by 5% since 1970, and is 2% lower than in 2005.

41%

have decreased in abundance.

More species have shown strong or moderate decreases in abundance (41%) than increases (26%) since 1970, and likewise more species have decreased in distribution (27%) than increased (21%) since 1970.

53%

of species show strong changes.

Our wildlife is undergoing rapid change; the proportion of species defined as showing strong changes in abundance, either increasing or decreasing, rose from 33% over the long term to 53% over the short term.

15%

of species are threatened.

Of 8,431 species that have been assessed using regional Red List criteria, 15% have been classified as threatened with extinction from Great Britain, and 2% are already extinct.

By 2020

most CBD targets won't be met.

An assessment based on the best available data indicates that, although progress has been made, the UK will not meet most of the CBD's 2020 Aichi targets.

The pressures

that have caused the net loss of biodiversity over recent decades continue to have a negative effect.

- Agricultural productivity, linked to the intensification of land management and the decline in farmland nature, is still increasing, although with government funding some farmers have adopted wildlife-friendly farming.
- Average UK temperatures have increased by nearly 1°C since the 1980s with widespread impacts on nature evident already.
- Legislation has driven marked reductions in emissions of some harmful pollutants, although negative impacts remain.
- Thousands of hectares of farmland, woodland and wetland are built on every year to meet the needs of our increasingly urbanised population, although woodland cover has increased, new wetland habitat has been created and heathlands and moors restored.



Photo: Ian Francis (rspb-images.com)

This report showcases a wide range of exciting conservation initiatives,

with partnerships delivering inspiring results to secure a brighter future for the UK's nature. Public support for conservation continues to grow, with non-governmental organisation (NGO) expenditure up by 24% since 2010/11 and a 46% increase in the time donated by volunteers since 2000. However, public sector expenditure on biodiversity, as a proportion of gross domestic product (GDP), has fallen by 42% since a peak in 2008/09, although the UK's expenditure on international biodiversity has grown.



Photo: Chris Gomersall (rspb-images.com)

The impacts of climate change and fishing

on species' abundance and distribution are evident throughout the UK's seas. At the base of the food web, plankton communities have changed in response to warming seas. While some fish stocks are showing signs of recovery, the impacts of decades of unsustainable fishing persist. The precise impact of other pressures on the marine environment, such as noise and plastic pollution, remain unclear.



Photo: Ben Andrew (rspb-images.com)

The UK has a long history of love for, and fascination with, its natural heritage.

Thanks to this, tens of thousands of volunteers collect data on wildlife every year. Without their dedication this report would not be possible; we thank them all.

#STATEOFNATURE

KEY FINDINGS

We are able to present trends in status for more species than ever before in *State of Nature 2019*. This is due to new datasets becoming available and the development of analytical tools which enable a much broader range of taxonomic groups to be represented.

Using multispecies indicators, our goal is to communicate a clear, objective assessment of the state of biodiversity in the UK. The metrics we present show how measures of average species' status have changed over time as well as showing the variation in trends between species.

We focus on measuring change over two periods: the long term, over nearly 50 years, and the short term, the last 10 years.

Our species' status metrics make use of two broad types of data:

Abundance data from a number of well-established monitoring schemes in the UK encompassing c700 species (birds, mammals, butterflies and moths). Many of these species are popular to record and are relatively easy to identify and to observe, making it possible to count individuals to get a measure of relative abundance. Our abundance metrics report the average change in relative abundance across these species.

Occupancy data from large-scale datasets, which we can now use to generate trends for 6,654 species across a wide range of taxonomic groups (including vascular plants, lichens, bryophytes and a number of invertebrate groups). These trends measure the change in the proportion of occupied sites, so our metrics effectively report the average change in range for these species.

We show trends for species for our long-term period, from 1970 to 2016 for abundance data and from 1970 to 2015 for occupancy data. Our short-term period covers the final 10-year period of these time series.

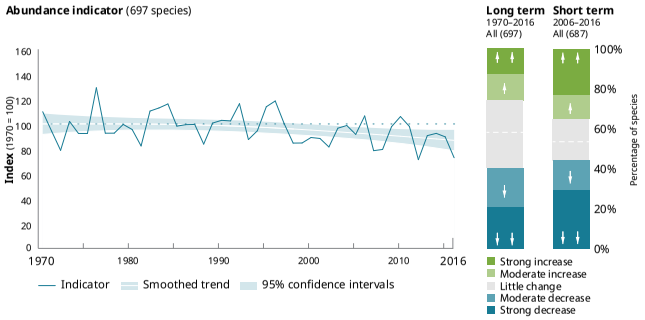
FIND OUT MORE

p92 How to interpret this report

Photo: Mike Read (rspb-images.com)



CHANGE IN SPECIES' ABUNDANCE



The abundance indicator for 697 terrestrial and freshwater species shows a statistically significant decline in average abundance of 13% (95% confidence intervals (CI) -22% to -5%) between 1970 and 2016. Over this long-term period, the smoothed indicator fell by 0.31% per year. Over our short-term period, the decline was a statistically non-significant 6%, a rate of 0.65% per year. There was, however, no significant difference in the rate of change between the long and the short term.

The white line with shading shows the smoothed trend and associated 95% CI, the blue line shows the underlying unsmoothed indicator. The bar chart shows the percentage of species within the indicator that have increased, decreased (moderately or strongly) or shown little change in abundance.

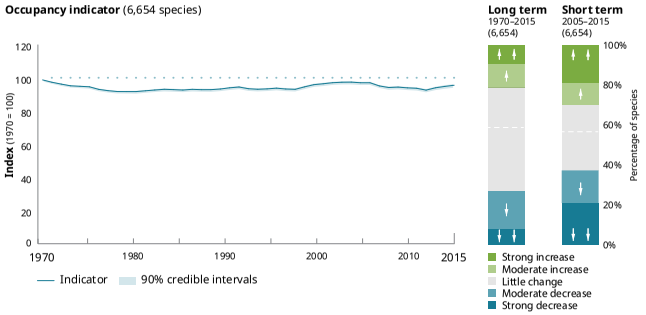
Within multispecies indicators like these there is substantial variation between individual species' trends. To examine this, we have allocated species into trend categories based on the magnitude of population change.

- Over the long term, 41% of species had strong or moderate decreases and 26% had strong or moderate increases; 33% showed little change.
- Over the short term, 44% of species had strong or moderate decreases and 36% had strong or moderate increases; 21% showed little change.
- Over the long term, 33% of species showed a strong change in abundance (either increase or decrease). Over the short term this rose to 53% of species.

Using a different, binary categorisation:

- Over the long term, 58% of species showed negative trends and 42% showed positive trends; over the short term, 53% of species showed negative trends and 47% showed positive trends.

CHANGE IN SPECIES' DISTRIBUTION



The occupancy indicator for 6,654 terrestrial and freshwater species shows that on average species' distributions have declined by 5% between 1970 and 2015. In 2015 the indicator was 2% lower than in 2005. Because species tend to decline in abundance before they disappear from a site, this change of 5% could reflect more severe underlying abundance declines that we are currently unable to quantify.

To examine the variation in species' distribution trends, we allocated species into categories based on the magnitude of distribution change.

- Over the long term, 27% of species showed strong or moderate decreases and 21% showed strong or moderate increases; 52% showed little change.
- Over the short term, 37% of species showed strong or moderate decreases and 30% showed strong or moderate increases; 33% showed little change.
- Over the long term, 17% of species showed a strong change in distribution (either increase or decrease). Over the short term this rose to 39% of species.

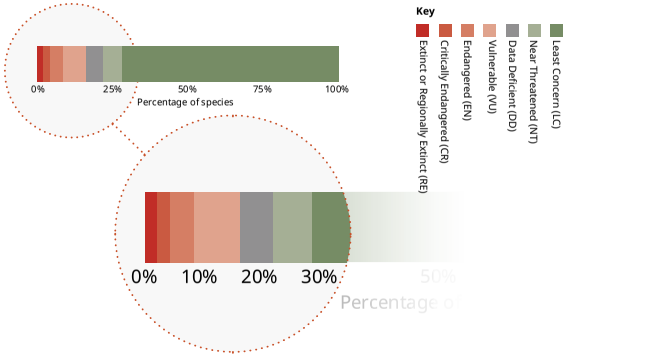
Using a different, binary categorisation:

- Over the long term, 58% of species showed negative trends and 42% showed positive trends; over the short term, 56% of species showed negative trends and 44% showed positive trends.

NATIONAL RED LIST ASSESSMENT

Here we show the percentage of species in Great Britain that have been allocated into each of the International Union for the Conservation of Nature (IUCN) Red List categories. Species assessed as Critically Endangered, Endangered or Vulnerable are formally classified as threatened and therefore at risk of extinction.

Percentage of species threatened = (CR + EN + VU)/(total number assessed – DD – RE).



Of 8,431 species that have been assessed against the IUCN Regional Red List criteria, 1,188 (15%) of the extant species for which sufficient data are available, are classified as threatened and therefore at risk of extinction from Great Britain. In addition, 2% of species are known (133) or considered likely (29) to have gone extinct since 1500, and a further four are extinct in the wild. Most extinctions in Great Britain are historic (1800 to mid-20th century), but losses continue; e.g. two breeding birds (Wryneck and Serin) have been lost this century. Several other species have not been found during recent targeted surveys and will almost certainly be classified as extinct in Great Britain when assessments are updated. An assessment of threatened species in Northern Ireland is given on page 70.

MARINE

Trends in abundance for marine species are shown by broad taxonomic group (see pages 94–97 for more details).

Demersal fish indicators show increases in average abundance in the Celtic and Greater North Seas of 133% and 58% respectively between the early 1980s and 2017.

The UK Breeding Seabird Indicator shows a 22% decline in average abundance for 13 species between 1985 and 2015.

Trends in the abundance of marine mammals, for which we have data, vary by group; cetacean species show stable populations since the early 1990s, and Grey Seal numbers continue to increase while Harbour Seal numbers are decreasing in a number of areas.

Changes in plankton communities are evident across the northern North Sea and the English Channel; the indicator for large copepods shows a 5% increase in the northern North Sea over the last 10 years, compared to a 41% decrease in the English Channel.

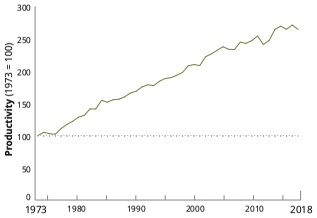
KEY FINDINGS — PRESSURES AND RESPONSES

As well as reporting on the state of the UK's nature, we have sought to identify trends in the key pressures that have caused the net loss of nature, as well as in the responses being made to help nature.

PRESSURES ON NATURE

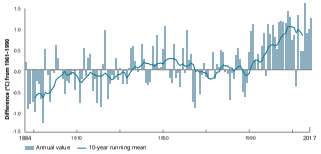
The *State of Nature 2019* reviews the major pressures on the UK's nature; agricultural management, climate change, urbanisation, pollution, hydrological change, invasive non-native species (INNS) and woodland management. Although monitoring change in these pressures, and the impact they have on the UK's biodiversity, has not been systematic, we have sought the most relevant available metrics to convey how the pressures on wildlife are changing.

Agricultural management



The steady increase in agricultural productivity, a product of increased land and resource use efficiency[1], is linked to the intensification of land management, which has had many documented impacts on the farmland wildlife found on 75% of the UK's land area.

Climate change



Climate change is driving widespread changes in the abundance, distribution and ecology of the UK's wildlife, and will continue to do so for decades or even centuries to come. The 21st century so far has been warmer than the previous three centuries. Since the 1980s, average UK temperatures have increased by **nearly 1°C**.

Pollution: emissions of many pollutants have been reduced dramatically in recent decades, although diffuse air and water pollution continues to have a severe impact on the UK's sensitive habitats and freshwaters, and new pollutant threats continue to emerge.

Urbanisation: there was an **8%** increase in proportion of UK population living in urban areas between 1970 and 2018.

Woodland management: woodland cover in the UK increased by **9%** between 1998 and 2018, although only **44%** of woodland is managed sustainably.

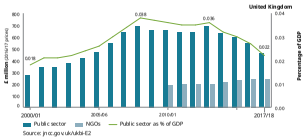
Hydrological change: 1,000 hectares (ha) of UK wetlands were converted to artificial surfaces in six years (2006–2012), although new habitat has been created, for example through post-mineral extraction restoration.

Invasive non-native species: On average **10–12** new non-native species become established in the UK annually, and 10–20% of these cause serious adverse impacts.

RESPONSE FOR NATURE

Conservation in the UK is delivered by a broad coalition of national and local governments, NGOs, businesses, landowners, farmers and private individuals, often working in partnership. There are many notable successes, some highlighted in this report, but as our state metrics demonstrate these have been insufficient as yet to stem or reverse the net loss of nature.

In recent years public sector expenditure on biodiversity conservation in the UK has declined (see figure below), both in absolute terms and as a percentage of GDP – the latter by **42%**, from 0.038% to 0.022%, between 2008/09 and 2017/18. Governmental expenditure on international biodiversity conservation, including in the UK's Overseas Territories (OTs), has increased steadily since 2000/01. A measure of NGO expenditure on biodiversity has increased since 2010/11, indicating growing public support for nature conservation.



Millions of people across the UK love and care for wildlife, as shown by support for wildlife charities, garden bird feeding and television viewing figures. Many show their support by donating time as conservation volunteers; the measure of this has **increased by 46%** since 2000. We estimate that around **7,500,000 volunteer hours** go into collecting biodiversity monitoring data every year.

Exciting and ambitious projects and policy commitments highlighted throughout the report illustrate the wide range of responses for nature in the UK and OTs, including but of course not limited to:

- Nature-friendly farming.
- Climate change mitigation schemes.
- Legally binding limits for pollution emissions.
- Habitat restoration schemes.
- Vast landscape-scale conservation projects to restore ecosystems.
- Community engagement and citizen science projects.
- Reintroduction schemes returning once-lost species.
- Managing problematic non-native species.

Continued monitoring of biodiversity will enable us to determine whether they are delivering the hoped-for recovery in the UK's wildlife.

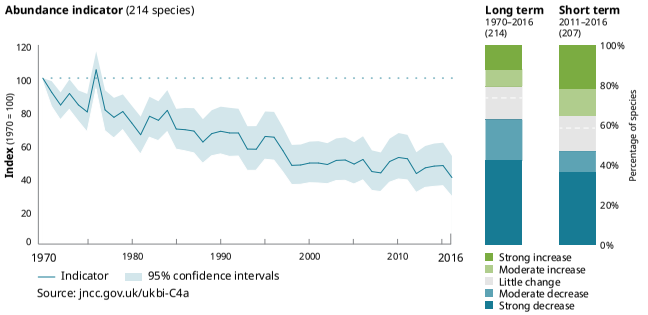
STATUS OF UK PRIORITY SPECIES

The official UK Biodiversity Indicators include two indicators to assess the status of species of greatest conservation concern; the UK's priority species. Of the 2,890 species included as priority species on the combined list for the four UK countries, data are available to assess change in relative abundance for 214 species and change in distribution for 395 species. In addition to the Priority Species Indicator, official UK Biodiversity Indicators are published annually for species groups including birds, butterflies and mammals, as well as other measures of biodiversity status.

We feature these indicators throughout *State of Nature 2019*. Note that the short-term assessments for UK Biodiversity Indicators are over five years, not 10 years as for the State of Nature indicators.

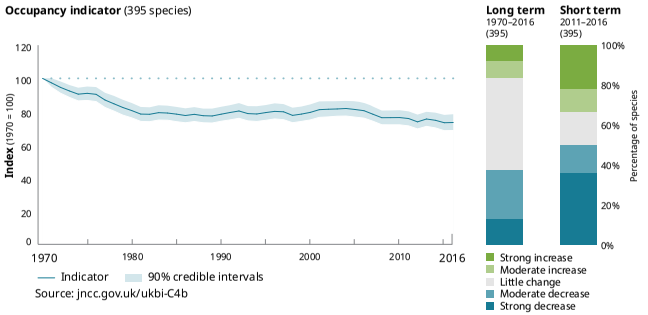
jncc.gov.uk/ukbi

UK Biodiversity Indicator: Change in the relative abundance of UK priority species, 1970 to 2016



- Since 1970, the indicator of abundance for 214 priority species has declined by a statistically significant 60%, and between 2011 and 2016 by 22%.
- Over the long term, 63% of species showed strong or moderate decreases and 22% showed strong or moderate increases; 16% showed little change.
- Over the short term, between 2011 and 2016, 46% of species showed strong or moderate decreases and 35% showed strong or moderate increases; 18% showed little change.

UK Biodiversity Indicator: Change in the distribution of UK priority species, 1970 to 2016



- Between 1970 and 2016, the index of distribution of priority species in the UK declined by 27%. The index was 3% lower in 2016 than in 2011.
- Over the long term, 37% of species showed strong or moderate decreases and 16% showed strong or moderate increases; 46% showed little change.
- Over the short term, between 2011 and 2016, 50% of species showed strong or moderate decreases and 33% showed strong or moderate increases; 17% showed little change.

Photo: Richard Brooks (rspb-images.com)


Bittern

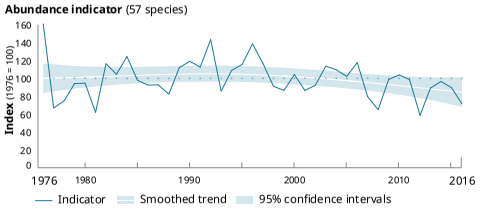
RESULTS IN MORE DETAIL

The composite nature of multispecies indicators means they do not show the variation in trends between individual species and species groups. In order to help understand changes in our *State of Nature 2019* headline abundance and distribution indicators better, we present indicators disaggregated into species groups.

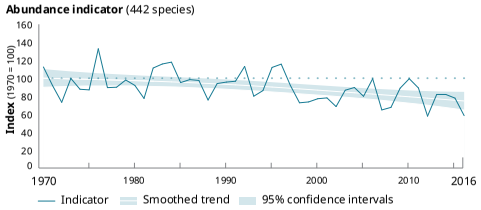
In the graphs shown here, the white line with shading shows the smoothed trend and associated 95% CI, the blue line shows the underlying unsmoothed indicator.

CHANGE IN AVERAGE SPECIES' ABUNDANCE BY TAXONOMIC GROUP

Butterflies, 1976 to 2016



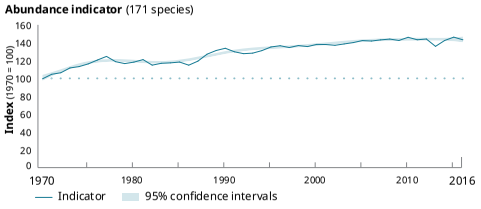
Moths, 1970 to 2016



The long-term decreases in average abundance in butterflies (-16%, CI -32% to 1%) and moths (-25%, CI -35% to -15%) have not slowed; short-term declines are 12% and 9% respectively.

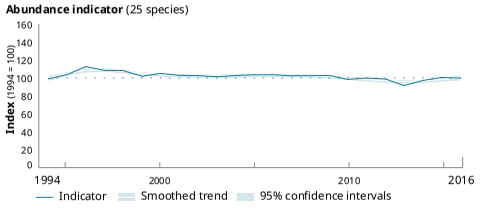
The UK Biodiversity Indicator for habitat specialist butterflies (not shown here; jncc.gov.uk/ukbi-C6) showed an unsmoothed decline of 68% between 1976 and 2018, showing a pattern of greater declines among habitat specialist species than generalists that appears to be prevalent in many taxonomic groups.

Terrestrial and freshwater breeding and wintering birds, 1970 to 2016



The abundance indicator for terrestrial and freshwater breeding and wintering birds shows a 43% (95% CI 39% to 47%) increase in average abundance over the long term but little change over the short term. The increase in this indicator is driven by recovery of some species from very low numbers, the arrival of colonising species, and increasing numbers of wintering waterbirds. These increases mask abundance declines in common and widespread breeding species: it is estimated that the total number of breeding birds in the UK fell by 44 million between 1967 and 2009[2].

Mammals, 1994 to 2016

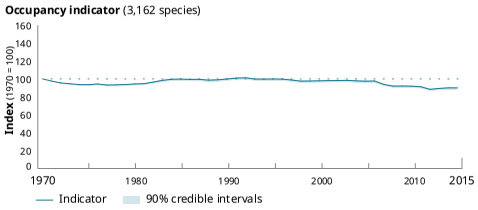


The abundance indicator for mammals starts in 1994 and shows little long-term change in average abundance. Over the short term, the indicator remained stable, just 3% lower in 2016 compared to 2006. Species trends vary and just over two-thirds of species show strong or moderate increases in abundance, some from very low baselines.

CHANGE IN AVERAGE SPECIES' DISTRIBUTION BY TAXONOMIC GROUP

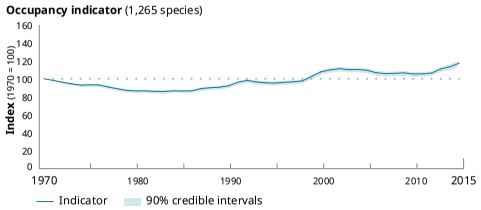
Similarly, the occupancy indicators can be separated to show the change in average species' distribution between 1970 and 2015 for four terrestrial and freshwater groups: insects, bryophytes and lichens, vascular plants and mammals. Some mammal and moth species are included in both our abundance and occupancy indicator, but a number are only included in the latter (see methods).

Insects, 1970 to 2015



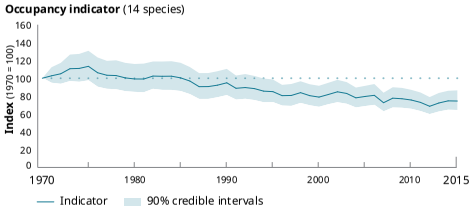
The occupancy indicator for insects shows a decrease in average distribution of 10% over the long term, and 8% over the short term.

Bryophytes and Lichens, 1970 to 2015



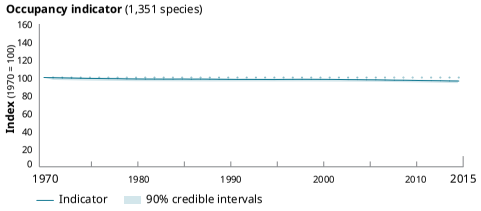
Over the long term the occupancy indicator for bryophytes and lichens shows a 17% increase in average distribution, having decreased during the 1970s and 80s. The indicator shows an 7% increase over the short term.

Mammals, 1970 to 2015



Over the long term the average distribution of mammals has decreased by 26% and is 6% lower over the short term.

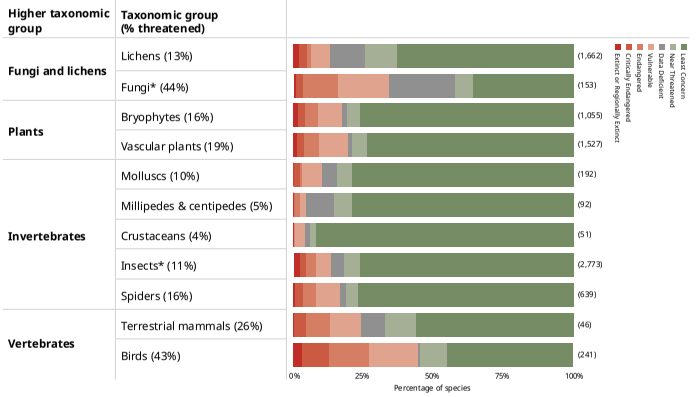
Vascular plants, 1970 to 2015



The occupancy indicator for vascular plants is 4% lower compared to 1970, and shows little short-term change in average distribution.

GREAT BRITAIN RED LIST ASSESSMENT BY TAXONOMIC GROUP

The bars show the percentage of assessed species, in broad taxonomic groups, falling into each of the IUCN Red List categories. Species assessed as Critically Endangered, Endangered or Vulnerable are formally classified as threatened and therefore at risk of extinction. The number of species assessed is shown in brackets.



*1% of fungi and 12% of insects were assessed.

Summarising these results by the main higher taxonomic groups, 440 plants (18%), 232 fungi and lichens (15%), 111 vertebrates (40%) and 405 invertebrates (12%) are classified as being at risk of extinction from Great Britain.

In addition, there are species in each group that are known to have gone extinct from Great Britain already: 32 plants (1.2%), 33 fungi and lichens (1.8%), seven vertebrates (2.4%) and 61 invertebrates (1.6%).

A breakdown of all-Ireland assessments for species in Northern Ireland is given on page 70.

HISTORICAL CHANGE IN BIODIVERSITY

***State of Nature 2019* focuses on recent changes in biodiversity, and the drivers of these changes, but we must remember that we have been shaping our landscape, and the wildlife within it, for millennia. It is widely accepted that the UK's biodiversity had been massively depleted by centuries of habitat loss, management changes, development and persecution before *State of Nature's* 1970 baseline. We are unable to measure this depletion accurately, but know many of the significant changes which occurred over the last two hundred years.**













THE GOOD NEWS

- In **1941**, Avocets return to the UK after a 100-year absence.
- Polecats begin a slow recovery in Wales from a low point in the **1930s**.
- The Large Blue butterfly is reintroduced to the West Country in **1985**.
- Salmon return to the Thames after a 125-year absence.
- Fisheries management enables the recovery of Herring stocks.
- Otters returned to every county in the UK by **2011**.
- By **2014**, the Lady's Slipper Orchid was flowering at 11 reintroduction sites.
- Corncrakes return to breed in Northern Ireland in **2016**.

ACTION TO HELP NATURE

- The first nature reserve was established in **1821** at Walton Hall, West Yorkshire.
- Wild Birds Protection Act 1876 introduced.
- The RSPB was formed in 1889, the National Trust in **1895** and the first of the Wildlife Trusts in **1912**.
- UK Government's Nature Conservancy established in **1949**.
- First National Park, the Peak District, designated in **1951**.
- Whaling by the UK ended in **1963**.
- Wildlife and Countryside Act **1981** introduced.
- The harmful pesticide DDT was banned in the UK in **1984**.
- Countryside Stewardship Scheme piloted in **1991**.
- The EU Water Framework Directive, addressing water pollution, comes into force in **2000**.
- The global 2020 Aichi targets are adopted in **2010**.
- In **2017**, UK carbon emissions drop to 43% below 1990 levels.
- Beinn Eighe in the Scottish Highlands becomes the UK's first Gene Conservation Unit protecting local Scots Pine lineages.

PRESSURES ON NATURE

- The first industrial revolution spanned **1780–1830**.
- 1,000km^2 of wetlands were drained annually between **1840** and **1880**.
- The UK's human population exceeded 30 million in **1871**.
- The introduction of steam trawlers caused a rapid increase in fishing during the **1880s**.
- The UK's first full-length motorway, the M1, opened in **1959**.
- 97% of wildflower meadows were lost between the **1930s** and **1984**.
- The UK joined the Common Agricultural Policy in **1973**.
- 10,000km^2 of land were drained in the **1970s**.
- The Central England temperature time series was 1°C warmer in latest decade compared to the pre-industrial period (**1850–1900**).
- The area of crops treated with pesticides increased by 53% between **1990** and **2010**.
- In **2019**, the UK's sixth national report to CBD indicates that the country is on track to meet five of the 20 Aichi targets by **2020**.

THE BAD NEWS

- The Great Auk was hunted to extinction in the UK in **1840**.
- Salmon disappear from the Thames in **1833**.
- Invertebrate extinctions hit a high in England, with 12 species lost between **1900 and 1910**.
- The loss of Mitten's Beardless Moss from Sussex in **1920** means the species goes extinct globally.
- Overfishing led North Sea Herring stocks to decline by over 99% between the **1960s** and **mid-70s**.
- Since the **1950s** wildflowers have been lost at a rate of up to nearly one species per year per county.
- Thirteen species of farmland bird were red-listed as Birds of Conservation Concern in **1996**, including Turtle Dove, Grey Partridge and Corn Bunting.
- The Freshwater Pearl Mussel became extinct from two Scottish rivers per year on average, between **1970** and **1998**.
- The indicator of habitat specialist butterflies down by 68% since **1976**.
- The Birds of Conservation Concern Red List increased from 36 to 67 species between **1996** and **2015**.

Photos: istock



DRIVERS OF CHANGE

We have identified the most significant pressures acting on terrestrial and freshwater nature in the UK: agricultural management, climate change, urbanisation, pollution, hydrological change, INNS and woodland management. Here we examine how each of these proximate drivers affects nature, the state of nature as a consequence of this, and the conservation actions being taken in response. We examine marine drivers of change on pages 50–63.

In the *State of Nature 2016* we reported the most significant drivers of change that have acted on the UK's wildlife since the 1970s, and attempted to quantify the impact these drivers had over that period, both to the benefit and detriment of nature. This research[1] identified that changing agricultural management had the biggest single impact upon nature in the UK over recent decades, with the great majority of that impact being to drive species' populations downwards. The second most significant driver was climate change, which is causing range and population change in sensitive species, alongside landscape-scale alteration to vulnerable habitats. Other important impacts included hydrological change, urbanisation and how woodlands are managed. Positive impacts were detected from wildlife-friendly farming, habitat management and the creation of new wildlife habitats.

Informed by this analysis, in the following pages of *State of Nature 2019* we take a more detailed look at what we believe to be the biggest threats to terrestrial and freshwater nature in the UK currently: agricultural management, climate change, urbanisation, pollution, hydrological change, INNS and woodland management.

Of course, in addition to these pressures, wildlife in the UK benefits from conservation, whether it is targeted action to save threatened species or the everyday actions of farmers and other land managers working to help nature across the country. We finish this section of the report by giving a brief overview of conservation in the UK, and how governments, NGOs, businesses, landowners, farmers and private individuals work together to save nature, showcasing a broad range of the many efforts being taken.

Photo: Mike Read (rspb-images.com)

AGRICULTURAL MANAGEMENT

A wide range of changes in agricultural management in recent decades has led to greater food production but they have also had a dramatic impact on farmland biodiversity. For example, populations of farmland birds have more than halved on average since 1970, and similar declines have been seen in many other taxonomic groups. Targeted wildlife-friendly farming, supported by government-funded agri-environment schemes (AES), can halt and reverse these declines, but to date the only successes have been for rare and localised species. The area of land receiving effective agri-environment measures may have helped slow the decline in nature but has been insufficient to halt and reverse this trend.

PRESSURE

Agricultural productivity has increased by over 150% since 1973.



STATE

Farmland bird indicator has fallen by 54% since 1970.



RESPONSE

Area under agri-environment increased to around 3 million ha.

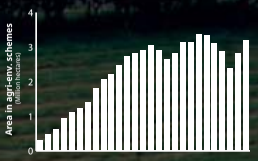


Photo: Andy Hay (rspb-images.com)

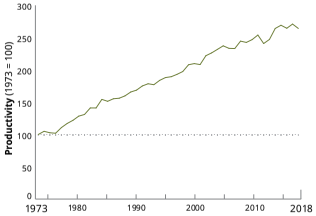
PRESSURES ON NATURE

Agriculture has been the dominant use of land in the UK for centuries, driven by the need to produce food for subsistence or profit since humans moved from hunter-gatherer societies to begin cultivating crops and raising animals. These practices have profoundly shaped historical and cultural perspectives on our landscapes and nature, and continue to do so today. Agricultural change has been identified as the most important driver of biodiversity change over the past 45 years[1], with most effects being negative. There are, however, also a range of species and habitats that largely depend on agricultural management.

Currently, 72% of the UK's land area is managed for agriculture, about one-third arable and two-thirds pastoral (grassland, moor and heath). Half of the arable land is used for cereal crops, while pastoral land is predominantly used to raise sheep (over 30 million) and cattle (over 10 million)[2].

Although historical changes have had massive impacts, it is only since the systematic recording of a suite of wildlife taxa began in the 1970s that we have been able to clearly link specific changes in management to changes in biodiversity. The changes in farmland management over the past 50 years that have had the greatest impact on the UK's nature include the increased use of pesticides and fertilisers; increased stocking rates, changes in crops and cropping patterns (e.g. grasslands managed for silage rather than hay production, with reseeding and drainage, crops sown in the autumn rather than the spring); farm specialisation (e.g. in either arable or livestock enterprises); greater mechanisation and increase in farm size; and loss of nature-friendly features such as field margins, hedgerows, wooded areas and farm ponds[3,4]. Over this period, agriculture has followed a consistent trend of increasing productivity (the ratio of inputs to outputs, a product of increased land and resource use efficiency), with associated consequences for wildlife. Of course, increased productivity does not of itself impact wildlife; it is some of the changes in management that have delivered increased productivity that have had a detrimental effect.

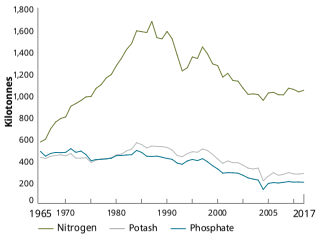
Agricultural productivity in the UK, a measure of intensification, 1973 to 2018[5]



An increasing awareness of the impact of modern farming methods on nature has led to changes in how public funds are used to support the agricultural sector. Since the 1990s a move away from direct production subsidies to area payments, coupled with requirements to meet basic environmental standards (cross-compliance) and the introduction of agri-environment schemes (AES), has aimed to mitigate some of the impacts of farming and help wildlife recover.

Although agricultural productivity continues to increase, the use of fertilisers, particularly nitrogen and phosphates, has decreased since peaking in the 1980s[6]. Numbers of sheep and cattle peaked in the 1970s and 1980s as a result of market trends and Common Agricultural Policy support payments, but have now fallen back. Spring-sown cereals, which can benefit farmland wildlife by providing an overwinter stubble, are also making a slight comeback, in part to combat herbicide-resistant weeds such as Black Grass.

Total quantities of nutrients used in the UK, 1965 to 2017[6]



Reported trends for pesticide use in the UK demonstrate some of the complexities involved in monitoring. Although the total weight of the active ingredient in pesticides has fallen markedly over the past 25 years, the number of hectares treated with pesticides, along with the frequency of treatments, have increased. In addition, there have been increases in the toxicity of pesticides and the variety of pesticides used on a single crop[7].

Trends in pesticide use in the UK, 1990 to 2016[8]

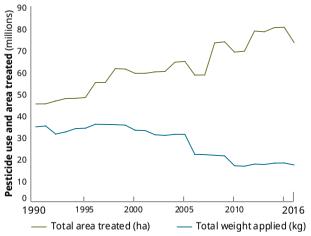


Photo: Colin Wilkinson (rspb-images.com)



THE STATE OF NATURE

One of the clearest examples of how farmland management has affected biodiversity is the trend in farmland birds; the suite of bird species most closely associated with farmland have declined more severely than birds in any other habitat, with a fall of 54% in the Farmland Bird Indicator since 1970[9]. Research has revealed the diverse processes by which species have been affected. For example, the shift to autumn sowing has resulted in a fall in Skylark breeding productivity as cereal crops become too tall and dense in the breeding season[10], and the loss of overwinter stubbles has meant poorer survival for granivores such as Yellowhammer[11]. Increased pesticide use has resulted in less invertebrate food for young Grey Partridges[12], while the drainage of wet grasslands and the loss of mixed farming systems has led to a decline in Lapwings[13].

FARMLAND BIRDS HAVE DECLINED MORE SEVERELY THAN THOSE IN ANY OTHER HABITAT.

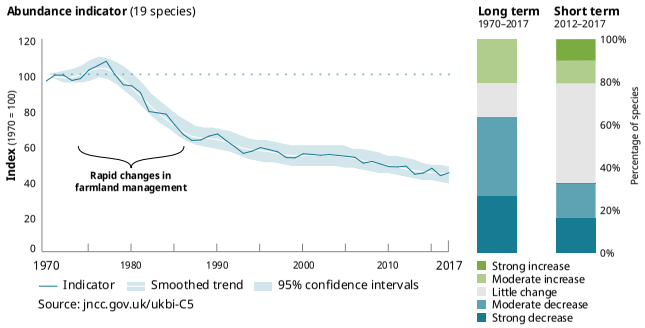
Trends do vary across the UK countries and generally farmland birds are faring better in Scotland, where on average they have increased since the 1990s, than elsewhere. Upland farming has arguably seen less dramatic changes than the lowlands, with some formerly widespread species now restricted to an upland range. However, the recent trends in upland birds show multispecies declines are occurring in these habitats as well[14].

The impacts of management changes are not limited to those on birds. A similar pattern of declines is evident for butterflies in the wider countryside (figure below), though for other insect and invertebrate groups the available data are insufficient to derive broad trends. There is growing concern about pollinators, largely related to use of pesticides such as neonicotinoids (which have seen recent restrictions in permissible use) but also the decrease in plant diversity and flower-rich habitats[15].

Arable plants such as Shepherd's-needle, Corn Buttercup and Pheasant's-eye have shown significant declines attributed to the use of fertilisers and herbicides and a range of management practices that reduce the seed bank or survival through increased crop density and decreased crop diversity[16]. Fertiliser use and conversion to arable have contributed to the loss of 97% of wildflower meadows and other species-rich grasslands in the past century[17].

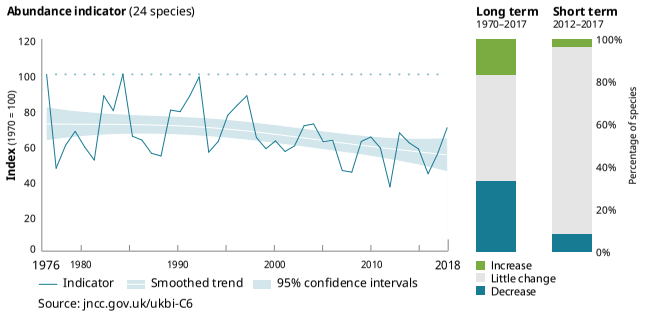
Farming has shaped the countryside for centuries and recent research shows that some current occupants including widespread bumblebees, brambles, Cow Parsley and Spear Thistle have adapted well to this landscape. Along with other adaptable generalists such as Woodpigeons and Jackdaws, some species appear resilient to agricultural intensification and have prospered in recent decades, while specialist species have seen widespread and often continuing decline.

UK Biodiversity Indicator: Trends in breeding farmland birds in the UK, 1970 to 2017



Source: jncc.gov.uk/ukbi-C5

UK Biodiversity Indicator: Insects of the wider countryside, 1976 to 2017 – butterflies

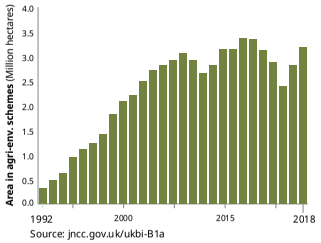


Source: jncc.gov.uk/ukbi-C6

THE RESPONSE FOR NATURE

One of the mechanisms for mitigating the negative impacts of agriculture is through environmentally sustainable farming practices. Although many farmers engage in nature-friendly farming voluntarily due to their own interests or as part of voluntary networks (for example, the Nature Friendly Farming Network and The Farming and Wildlife Advisory Group), government-sponsored AES have provided the main impetus. Although there are challenges to implementing particular actions at a sufficient scale to reverse declines in widespread species, AES provide a mechanism to target the most appropriate conservation actions in the relevant areas.

UK Biodiversity Indicator: Area of land in agri-environment schemes in the UK, 1992 to 2018



Source: jncc.gov.uk/ukbi-B1a

There has been widespread uptake by the farming community, but the effectiveness of AES has been hard to demonstrate. Despite the proven value of many AES options in field trials, a number of broader studies have failed to establish clear cause and effect for population recovery at larger geographical scales, largely as a result of the complex interactions of different factors. Where broad-scale successes have been identified for birds[18], these have been mainly related to the provision of winter food through stubble management and wild bird seed mix. Some moth and bat species have been shown to benefit, at a local scale, from the presence of field margins and boundary features that include mature trees[19,20]. There has, however, been little evidence of benefits to species occupying in-field cropped habitats[21].

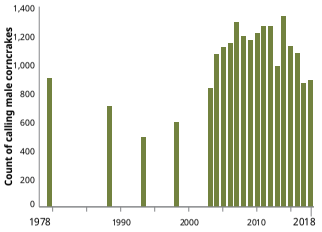
Although agricultural intensification may have slowed since its late 20th century peak, and there have been notable successes in recovering some threatened species such as Cirl Bunting and Stone Curlew, aggregate farmland biodiversity indicators continue to decline despite government commitments to reversing the downward trend and a huge effort from farmers and the conservation sector. This may in part reflect a lag in the response of wild populations to habitat change, meaning that declines can continue even after habitat change has halted or been partly reversed[22]. Research has identified many of the issues contributing to continuing population declines, but a key factor for some species is the lack of implementation of remedial action at a coordinated landscape scale sufficient to make a real difference.

CASE STUDIES

Corncrake recovery

Successes for individual species have been achieved where research has led to the identification of a particular problem and resources deployed at a sufficient scale to tackle it – typically for very rare species with a restricted range, such as the Corncrake in the Scottish islands. Here, farmers and crofters receive targeted advice regarding habitat management and mowing regimes, and receive payments to delay grass cutting. Male Corncrake numbers have risen from a low of under 500 in the early 1990s to 1,289 in 2014. Although this is a success, the population is still tiny with a highly restricted range and is no longer a breeding species on the Isle of Man. Setting this in context, the species was once found in all counties of the UK and in every meadow and cornfield in the north of Ireland, where one late 19th century author reported that "its incessant cry is monotonous if not wearisome"[23]. Since 2014, numbers in the core area fell for three consecutive years, with the 2017 count the lowest since 2003. Alongside unusual spring weather, a reduction in payments to delay mowing and a reduction targeted advice may have contributed to this downturn, highlighting the vulnerability of rare species to changes in policy.

Calling male Corncrakes in Great Britain and Isle of Man, in survey years 1978 to 2018



Land management for butterflies[24]

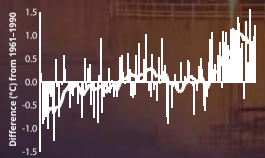
The Marsh Fritillary is one of the fastest declining butterflies in the UK, having lost two-thirds of its colonies between 1990 and 2000. The main cause is the loss of damp tussock-forming grassland, heath and mire that contains Devil's-bit Scabious, the plant that the caterpillars require for food. By working with landowners, Butterfly Conservation and Natural England helped secure Higher Level Stewardship for landholdings on Dartmoor where they were able to introduce butterfly-friendly measures such as controlled grazing, scrub removal, increased connectivity between patches and reintroduction of the larval food plant. By 2010, the area of occupied habitat had almost tripled, and in occupied areas, the counts of larval webs had risen tenfold. Facilitated by AES funding, the project has been a great success, due to strong partnership working between the project officer, Natural England advisors, Dartmoor National Park ecologists and the farmers and volunteers delivering the management. Ongoing management advice is crucial if benefits are to continue, and the follow-on project, All the Moor Butterflies, is producing similar positive results.

CLIMATE CHANGE

Climate change, caused by human activities, is one of the most significant threats to global biodiversity, and is projected to become increasingly severe through the course of this century[1]. While climate change has had the second largest impact (after agricultural change) on UK nature over the last 40 years, impacts on wildlife have been mixed[2]. There is growing evidence that climate change is driving widespread and rapid changes in the abundance, distribution and ecology of the UK's wildlife, causing changes to species communities and will continue to do so for decades or even centuries to come. Conserving and restoring nature-rich areas of the UK will contribute to mitigating climate change and benefit species, while strategies to counter the negative effects of climate change will help species to adapt to its increasing influence in future.

PRESSURE

All the top 10 warmest years since records began have occurred post-1990.



STATE

Climate change has already impacted population trends of moths (explaining 40% of the decline) and aphids (explaining 60% of the increase).

RESPONSE

Protecting nature-rich areas secures carbon, benefits species and provides vital ecosystem services – providing benefits all round.

Photo: Peter Cairns (rspb-images.com)

PRESSURES ON NATURE

Increasing temperatures and rising sea levels are just two symptoms of the complex climatic changes that have been apparent in the UK over recent decades.

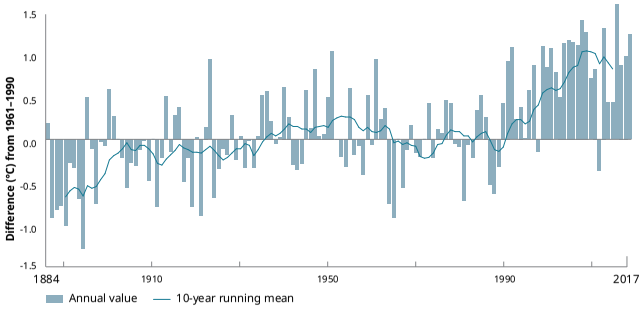
The graph to the right shows annual mean temperature for the UK, expressed as anomalies relative to the 1961–1990 average. The blue line is a smoothed trend, which is roughly equivalent to a 10-year running mean.

The UK has experienced significant warming, with mean annual temperature in the most recent decade (2008–2017) nearly 1°C warmer than the 1961–1990 average. These trends are apparent in all countries, and across all seasons, with spring warming being the most pronounced (1.1°C warming). There has been a reduction in the number of days of ground frost and an increase in the number of growing days. There has also been an increase in the amount of sunshine by 6% from 1961–1990 to 2008–2017[3].

Rainfall shows more variability between years, making trends more difficult to spot. However, the most recent decade has been 8% wetter than the 1961–1990 average, with trends being most apparent in winter and summer.

Climate change is also affecting conditions at sea; sea surface temperature has increased by 0.6°C from the 1961–1990 average and the sea level has risen by 16cm since the start of the 20th century. Sea-level rise increases pressure on intertidal habitats and may lead to more severe flooding and coastal erosion by waves[4].

Time series of average UK land temperatures in °C since 1884, expressed as anomalies relative to the 1961 to 1990 average[3]



PROJECTIONS FOR 21ST CENTURY CLIMATE

The current predictions of the future UK climate, published most recently in November 2018, are provided by the UK Climate Projections[5,6]. Projections are made for several different emissions scenarios. Given current levels of emissions, we present those for a high emissions scenario.

There is projected to be a greater chance of warmer, wetter winters and hotter, drier summers and by mid-century the chance of hot summers, resembling the 2018 heatwave, will increase from <10% to 10–20%. Rainfall patterns will change significantly; increasing in winter and decreasing in summer. Sea-level rise will occur for all emission scenarios and at all locations around the UK with the associated risk of increased coastal flooding. Extreme weather events, such as drought and flood, will increase in frequency.

20th century climate change – most recent decade 2008–2017 compared to 1961–1990 average

	Temperature	Rainfall
ON LAND	1°C warmer	8% increase

	Temperature	Sea level
AT SEA	0.6°C warmer	16cm rise since start of 20th century

Projections for 21st century climate: by 2070

	Temperature	Rainfall
WINTER	0.7 to 4.2°C warmer	-1% to +35%
SUMMER	0.9 to 5.4°C warmer	-47% to +2%

	Sea surface temperature	Sea level
AT SEA	1.5 to 4°C warmer	c30cm to115cm*

*Relative to 1981–2000.

THE STATE OF NATURE

Temperature, rainfall and other climatic factors consistent with a warming climate affect the abundance and occurrence of individual species, acting on range, population and phenology (the timing of seasonal events)[7,8,9].

DISTRIBUTIONAL CHANGES



Photo: Jeroen Stel (rspb-images.com)

In the UK many species, including birds, butterflies, moths and dragonflies, have moved north over the last four decades. Shifts for these groups averaged 23km per decade between the 1970s and 1990s, and 18km per decade between the 1990s and the mid-2000s[10]. In the marine environment, warming seas have led to changes in plankton and fish distribution resulting in changes to species composition. Such shifts in range rely on there being suitable habitat to permit movement and establishment, and on species' ability to disperse. The threat from climate change therefore interacts directly with other pressures on species, meaning that impacts can be exacerbated[11].

RELATED DRIVERS OF CHANGE

p54 Marine > Climate change

p18 Agricultural management

p26 Hydrological change

CHANGES IN TIMING

The onset of seasonal events is now earlier for a wide range of marine and terrestrial species. Species' responses vary: for terrestrial plants and invertebrates this has been by about four days on average for a 1°C increase in temperature, while for birds the figure is two days on average[12]. Different responses by different parts of the food chain may disrupt ecosystems if the availability of prey and the timing of peak predator requirements do not align. Although there are examples of this impacting particular populations[13] there is, at present, only weak evidence that this mismatch has driven observed long-term declines in predatory species, such as insectivorous birds[14].

POPULATION TRENDS



Photo: Tom Marshall (rspb-images.com)

Our *State of Nature* metrics show declines in moths and butterflies across the UK, although trends in Scotland are, on average, stable. A recent study modelling climate impacts on species' abundance concluded that across all moth species, approximately 40% of the significant decline could be attributed to climate change[15]. Increases in the abundance of flying aphids is probably due to warmer temperatures, which reduce generation time[16], leading to more aphid generations in a year.

There is growing evidence that warming has driven significant changes in plant communities in the Scottish Highlands, with declines in arctic-montane species[17], and in specialist snow-bed species[18]. The impacts of seasonal changes in climate vary across different taxa. Summer drought can have a significant impact on the growth and survival of tree species, leading to major changes in the composition and structure of woodland[19].

Whether an individual species is seen to benefit from, or to be adversely affected by, climate change within a particular geographic region, the resultant shifts in species' distributions will impact on the structure of communities. This can impact community processes and species' interactions (such as predator-prey relationships and competition) which can in turn drive further population change[20].

MANY SPECIES, BOTH ON LAND AND IN THE SEA, HAVE SHIFTED THEIR RANGES NORTHWARDS IN RECENT DECADES.

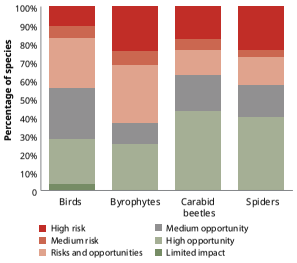
THE RESPONSE FOR NATURE

Responses to combat the negative effects of climate change on species involve planning and conservation action at a range of scales. It will be essential for us to understand the risks and identify areas, habitats and species groups that are most vulnerable.

ASSESSING THE RISKS TO SPECIES IN ORDER TO PRIORITISE RESOURCES

A detailed assessment accounting for potentially confounding and exacerbating factors applied to 402 species suggested that 35% are at risk of range loss, particularly among the bryophytes and vascular plants, while 42% may expand their range. Notably, three-quarters of the upland species considered are predicted to be at risk from climate change[21].

Example results for four taxonomic groups (birds, bryophytes, carabid beetles and spiders) from assessment of climate vulnerability for Great Britain across 17 taxa, separating those species regarded as negatively impacted (medium and high risk), from those with potential opportunities to benefit (medium and high opportunity)[21].



PROTECTING AND ENHANCING THE PROTECTED AREA NETWORK

There is strong evidence that protected areas remain a vital conservation tool in spite of shifting species' distributions. For eight invertebrate taxa, colonisations have occurred on protected sites more often than expected by chance based on the availability of protected areas[22]. More specifically, colonisations by seven butterfly and bird species are over four times more likely in protected areas than expected by chance[23,24] – this is probably because such sites better preserve important areas of semi-natural habitat. There is also some evidence that protected areas reduce the risk of population extinction towards the southern range-margin of species' distributions[24]. This may be because protected areas protect high-quality habitats or because conservation management at those sites reduces negative impacts of climate change. Notwithstanding this, areas projected to have suitable climates would need to be targeted for habitat restoration and creation to provide space for species to move into, and improved habitat connectivity is vital, particularly for less mobile species.

SITE-BASED MANAGEMENT

BogLIFE – Lowland raised bog sites[25]

The Cumbria BogLIFE project is undertaking large-scale restoration works, using specialist contractors and innovative techniques that are:

- Re-creating wetter, boggier ground – by blocking drainage ditches and creating "bunds" (little walls of undamaged, wet peat) that will hold water within the bog.
- Removing some areas of trees that have started to grow in the artificially dry conditions – as bogs do not naturally have trees growing across their surface.
- Reintroducing bog plants to areas with a bare peat surface. This has included growing trials of Sphagnum mosses to replace peat in gardening, with commercial potential across lowland peat agricultural soils.

Butterfly banks



Photo: Patrick Cashman (rspb-images.com)

At an ex-arable farmland site in Wiltshire, 200ha of species-rich grassland has been created for the benefit of specialist plants and butterflies. For butterfly species at the north of their climatic range in the UK, such as the Adonis Blue and Silver-spotted Skipper, varied topography with steep, warm south-facing slopes is essential. To support existing populations and increase connectivity in the landscape, butterfly banks have been constructed by bulldozing chalk up from the field into an S-shaped bank. This ingenious shape ensures that its sides face in all possible directions, providing warm slopes throughout the day and provides a variety of microclimates to support a wide variety of insects. A population of Small Blue butterflies has become established just two years after it was created.

Conserving carbon and conserving species

A recent project mapped the best places for nature across the UK and calculated the amount of carbon contained in the vegetation and top 30cm of soil[26]. Across these nature-rich areas, the amount of carbon, if lost to the atmosphere, would equate (very conservatively) to two gigatons of CO_2, equivalent to four years of the UK's annual overall CO_2 emissions. Only 34% of this carbon is currently protected. Poor habitat condition often means that stored carbon is being lost to the atmosphere – as has been well documented for peatlands[27]. Mapping is the first step to integrating these carbon and nature-rich areas into plans and initiatives to secure their protection, restoration and maintenance to ensure long-term benefits for both nature and people.

HYDROLOGICAL CHANGE

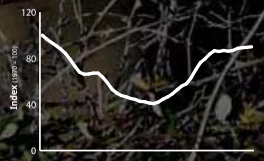
Throughout history humans have sought to alter natural water systems for their benefit – such that there are few pristine freshwater ecosystems remaining in the UK. Many rivers have been straightened, piped and dammed, marshland and agricultural land has been drained, groundwater abstracted and floodplains built upon. Species reliant on the range of wet habitats affected by these changes have seen long-term declines and face ongoing pressures of unsustainable water abstraction and the continuing drainage and conversion of wetlands to other land uses. Recent large-scale projects addressing some of these issues include the rewetting of upland peat bog that had been drained for agriculture and forestry; the re-creation of lowland wetland habitats; river restoration; and pond creation. Despite these landscape-scale projects, the extent of wetland habitats remains greatly reduced compared to the middle of the last century.

PRESSURE

In 2018, just 35% of water bodies had "good" or "high" ecological status.

STATE

Change in distribution of freshwater invertebrate species, 1970 to 2015.



RESPONSE

Since 2010, the Nature After Minerals partnership has created 2,000ha of wetland through the restoration of minerals sites across the UK.

Photo: Richard Brooks (rspb-images.com)

Marsh Harrier

PRESSURES ON NATURE

Many of the pressures affecting the distribution and quality of freshwater habitats relate to historical land drainage. Drainage of the great fens of East Anglia began in Roman times and continued with further major drainage in the 17th century. Technological innovations introduced during the 19th century transformed seasonally flooded marsh and pasture into the agronomically productive farmland we see today[1]. Field drainage has been a major technical factor in relieving modern farming from previous limitations on production. Improvements to farming productivity have simultaneously led to the loss of other ecosystem services including water regulation and soil stabilisation.

Drainage activity in post-Second World War Britain rose to a peak in the 1970s due to government grant payments – about 1 million ha was drained in that decade alone[2]. An estimated 300,000ha of lowland wet grassland were lost between 1970 and 1985[3], and 1.5 million ha of upland blanket peatland was drained in the mid-century, mainly through the mechanical digging of surface grips (channels designed to lower the water table)[4]. Upland drainage reduces carbon storage and can contribute to lowland flooding, with impacts on people and their livelihoods. The loss of wetlands continues to be a cause for concern; land-cover maps show that between 2006 and 2012 over 1,000ha of wetland was converted to artificial surfaces[5].

Alongside general land drainage, loss of specific wet features on farmland has been substantial. Up to 90% of lowland ponds in the UK were lost in the 20th century, through neglect or direct human intervention[6]. Those that remain face increasing pressure due to agricultural land drainage, pollution, isolation and urban development[7].

Rapid population growth in recent decades has led to an increase in demand for freshwater. This has been particularly apparent in South East and Eastern England where 22% of freshwater is abstracted. These regions are categorised by the Environment Agency as overexploited[8].

Some of the most acute problems of over-abstraction have occurred in lowland chalk stream systems, which are of global ecological importance. The Water Framework Directive (2000)[9] committed EU member states to achieving good qualitative and quantitative status of all water bodies (including marine waters up to one nautical mile from shore) by 2015. Of all of our waters, rivers are perhaps facing the most pressures. Despite the requirements of the WFD, there has been a slight decline in the percentage of water bodies with "good" or "high" ecological status, to 35% in 2018. Diffuse pollution from agriculture remains a key issue[10]. Physical changes to rivers, lakes and estuaries, such as flood defences and weirs, affect the ecology of over a third of our waters, preventing them from functioning naturally. Further pressures on the hydrological network come from climate change (both increasing flood risk and prolonged droughts) and INNS, which have had major impacts on freshwater ecosystems. These issues are discussed further in the relevant sections elsewhere in this report.

RELATED DRIVERS OF CHANGE

p34 Invasive non-native species, pests and pathogens

p38 Pollution

p22 Climate change

Photo: Richard Revels (rspb-images.com)

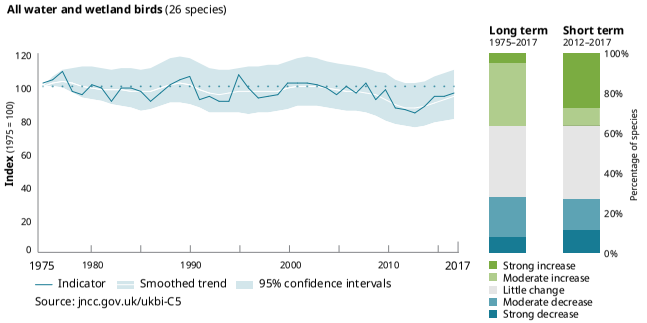


THE STATE OF NATURE

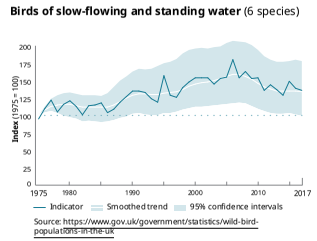
Changes in the populations of water and wetland birds can be closely linked to hydrological changes, although the impacts vary widely across species, and other factors interact and play their part. The breeding water and wetland bird indicator for the UK fell by 6% between 1975 and 2017, but over the short term has increased slightly, by 3%[11].

Within the indicator, species can be split into groups based on breeding habitat. Species associated with slow-flowing and standing water, and with reedbeds, have on average shown increasing trends, with a range of species benefiting from new habitat created through the restoration of gravel pits after extraction has finished and improvements in river management. Conversely, birds of fast-flowing (typically upland) rivers, and wet grasslands, have declined on average. Declines have been most notable in breeding waders of lowland wet grassland such as Lapwing and Snipe, due to habitat loss. Outside Scotland, a large proportion of these species' populations are now confined to sites managed as nature reserves.

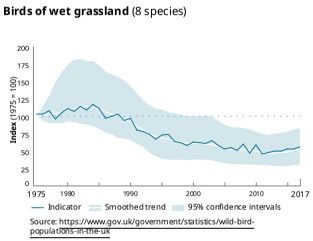
UK Biodiversity Indicator: Breeding water and wetland birds in the UK, 1975 to 2017 – Abundance indicators



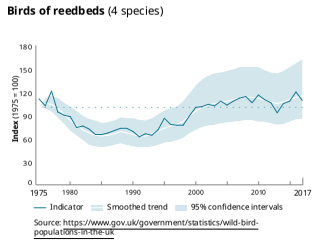
Source: jncc.gov.uk/ukbi-C5



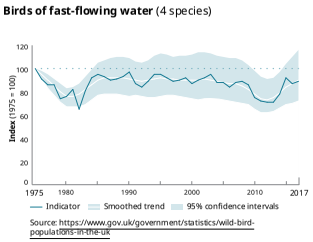
Source: https://www.gov.uk/government/statistics/wild-bird-populations-in-the-uk



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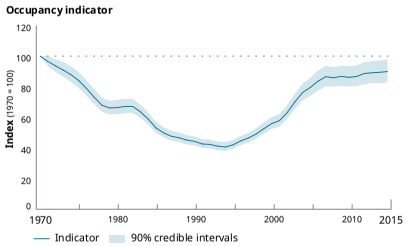


Source: https://www.gov.uk/government/statistics/wild-bird-populations-in-the-uk



Source: https://www.gov.uk/government/statistics/wild-bird-populations-in-the-uk

Change in distribution of freshwater invertebrate species, 1970 to 2015[12]



Occupancy modelling has revealed interesting patterns in the distribution of several groups of aquatic macroinvertebrates. Freshwater invertebrates, Stoneflies (*Plecoptera*), Caddisflies (*Trichoptera*), aquatic bugs (*Hemiptera*), Dragonflies (*Odonata*), Mayflies (*Ephemeroptera*) and freshwater molluscs were recorded at a declining number of sites through the 1970s and 1980s, followed by strong increases in the years since. The drivers for this recovery are not fully understood, and improvements in water quality (such as a recovery from acidification) are likely to have played an important role, but hydrological management to restore rivers, changing river flow conditions and the presence of specific local-scale habitat features may have played a part for some species[13,14,15]. In addition, concerns remain about the status of aquatic macroinvertebrates more generally, including the widespread impacts of sediment and pesticide run-off, as well as the localised impacts of pollution events.

THE RESPONSE FOR NATURE

Given that good hydrological management can bring major benefits for humans through the provision of clean water for consumption and industrial use, and flood prevention, many recent policy actions have primary aims in these areas. However, as good hydrological practices also benefit nature, this is increasingly recognised as having all-round benefits by policymakers.

CASE STUDIES

Creating new wetlands on mineral extraction sites



Photo: Kevin Harwood (rspb-images.com)

The restoration of worked-out sand and gravel extractions to wetlands has become widespread in recent decades and provides an important opportunity for increasing the extent of wetland habitats. Partnerships between conservation organisations and operating companies help to ensure success. For example, as part of the Nature After Minerals project, the RSPB, Natural England, the Mineral Products Association and the British Aggregates Association work alongside quarry companies and planners to create new spaces for nature at worked-out quarries. Transforming previously industrial sites to places where nature can thrive is a powerful symbol of what can be achieved through effective partnerships. Highlights of this approach include:

- Over 8,000ha of new habitat (including 2,000ha of wetland) created and managed since 2010.
- With specialised ecological advice, a range of habitat features can be created to benefit rare species.
- 13% of all the UK's breeding Bitterns now nest in restored mineral sites.
- When training is delivered with and for industry partners, best practice can be shared.

Examples of habitat creation include the UK's largest created reedbed through the Hanson–RSPB Wetland Project in Cambridgeshire and 1,000ha of new habitats created over 10 years through the CEMEX–RSPB partnership.

Bringing back the Eurasian Beaver



Photo: Nick Upton (rspb-images.com)

Eurasian Beavers are native to the UK, and were once widespread across Scotland, England and Wales, but were hunted to extinction by the end of the 16th century.

Beavers are well known for their dam-building habits and can be considered as ecosystem engineers, with the ability to rapidly alter the hydrology of the landscape they occupy. By blocking flows, they slow down the passage of water and create pools with diverse structures while modifying the local habitat through natural coppicing, opening of glades and creating deadwood.

Recent efforts to restore Beavers to Scotland were led by the Scottish Wildlife Trust and Royal Zoological Society of Scotland with Scottish Natural Heritage and Forestry Commission Scotland. As a trial, four families of Beavers were released at Knapdale in Argyll in 2009. An intensive programme of monitoring and research assessed the impact of the trial, and of an additional unauthorised release on the Tay at around the same time, on both the natural and the human environment[16]. Conclusions from this work include:

- Beavers have an overall positive influence on biodiversity with a wide range of species benefiting from the habitats created, including fish, amphibians and a wide range of invertebrates.
- Ecosystem services are provided, such as increased groundwater storage, water flow stabilisation and flood prevention.
- A number of species may be adversely impacted, including Aspen woodland lichens, bryophytes, fungi communities and some invertebrates[16].

In England, the five-year River Otter Beaver Trial is being led by Devon Wildlife Trust with research by the University of Exeter and is due to report in 2020. In Wales, plans for a trial are being developed by the Welsh Beaver Project.

The future for Beavers

- There is an estimated 120,000ha of "potential Beaver woodland" – appropriate broad-leaved woodland in suitable proximity to freshwater – in Scotland.
- In 2014, 84% of respondents in a survey of mid-Argyll residents were in favour of Beavers continuing to live in the area[17].
- The monetary value to society of the Knapdale trial has been estimated at up to £6.7 million with the monetary cost of civil engineering impacts and timber loss being put at no more than £44,000[18].
- The Scottish Government concluded in 2016 that Beavers were in Scotland to stay and in May 2019 they were granted European Protected Species status.

With an increasing number of Beavers in the wild and in fenced trial sites in England and Wales, and provided that the right balances can be struck, it may be that this charismatic ecosystem engineer is making a long-awaited return to the UK.

URBANISATION

The end result of urbanisation – the process of making an area more urban – can vary from extensive sprawl, where built land is interspersed with green spaces, to intensively urbanised areas where compact built environments are separated from green space. Urbanisation need not always result in biodiversity loss: the conversion of an intensively managed arable field to a housing estate with gardens, a community orchard and a pond may provide net gain for species diversity and abundance. Conversely, much valuable habitat, including brownfield and semi-natural habitat, has been lost or damaged by development. Management and habitat creation projects can improve habitat diversity, resource availability and connectivity across green spaces within urban environments. As well as helping biodiversity, these can enhance ecosystem function and help human well-being[1].

PRESSURE

There was an 8% increase in the proportion of the UK's population living in urban areas between 1970 and 2018.

STATE

Between 2006 and 2018, 1,600 miles of road were constructed in Great Britain.

RESPONSE

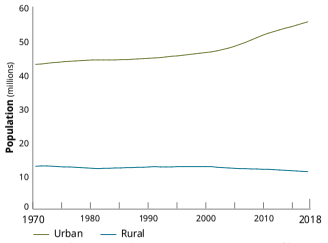
The concept of net gains for biodiversity through development have recently been enshrined in planning policy in England and Wales.

Photo: Andy Hay (rspb-images.com)

PRESSURES ON NATURE

The UK's human population has risen steadily over the last century, mainly centred around urban areas (see figure)[2]. This has required large-scale infrastructure developments, to meet demands for food, goods and human movement. While the increasing intensity of development within specific urban zones potentially reduces impacts on the wider countryside, the biodiversity value of existing urban green spaces and wildlife-rich natural areas can be impacted.

UK urban and rural population, 1970 to 2018



Source: https://population.un.org/wup/Country-Profiles/

Analyses presented in *State of Nature 2016* showed that urbanisation accounted for a greater impact on species than any other habitat conversion[3]. Comprehensive data on habitat loss to urbanisation are not available, but we know that an additional 1,600 miles of road were constructed in Great Britain between 2006 and 2018[4]. Permanent damage to soil function and increased diffuse pollution from urban run-off are two detrimental results of covering land with built, impermeable surfaces. In recognition of this, Scotland has recently introduced a new Environmental Health Indicator for sealed soil, which enables development to be tracked more robustly. Between 2009 and 2018, the area of sealed soil in the Clyde catchment, for example, increased from 4.38% to 5.2%, representing an area of 41,149ha[5].

Urbanisation has direct consequences for wildlife in terms of land use and land cover changes, but it also acts to fragment landscapes by creating barriers between habitats, thus isolating some populations and in turn reducing their genetic fitness[6]. A wide variety of green spaces exist within urban environments, the biodiversity value of which varies with the degree of fragmentation, management, local population density and surrounding land use. Increases in air, light and noise pollution, human disturbance and predation by domestic animals particularly affect biodiversity in urbanised areas.

Nevertheless, most urban environments are not completely covered with built structures but contain a substantial area made up of domestic gardens, parks, allotments, cemeteries, ponds, road verges and brownfield sites. Detailed analysis of four British urban centres (Bristol, Edinburgh, Reading and Leeds) show that over 60% of the total landcover is "green" – with residential gardens making up 24–36% of each city[7]. While much of this area is not managed for the benefit of nature, the value of green infrastructure for biodiversity is worthy of consideration and it is also important for the health and well-being of the human inhabitants.

Photo: David Broadbent (rspb-images.com)



THE STATE OF NATURE

DIRECT HABITAT LOSS

Development for housing, industry and infrastructure projects such as road and rail result in a loss of natural habitats, as well as fragmentation and change to those that remain. Heathland is one of the habitats that has been most impacted by urbanisation. The UK's remaining lowland heathland is distributed predominantly in southern England, and most is in close proximity to expanding urban centres. Direct loss, fragmentation and degradation through nitrogen deposition and disturbance has negatively impacted vulnerable species, including heathland reptiles such as Sand Lizard and Smooth Snake, and rare ground-nesting birds like Nightjar and Woodlark. Although direct loss of heathland habitat through land-use change is now controlled by planning and environmental legislation, it was still occurring until the 1990s[8]. Development in surrounding areas continues to increase the pressure on remaining isolated heathland patches, through increased fire risk, predation by domestic animals and human disturbance.

NATURE IN URBAN AREAS

Urban environments can support high levels of biodiversity and offer opportunities for some species. Low and intermediate levels of urbanisation can increase species' richness for some groups, which has been associated with the mosaic of habitats available[9]. Urban areas have been shown to support higher bee species' richness than non-urban habitats and a recent study found that residential gardens and allotments are pollinator hotspots[10]. Green infrastructure such as sustainable drainage systems can provide habitat for a wide range of species, including amphibians and invertebrates, but many sites do not fulfil their potential as wildlife habitats[11].

Hedgehogs, now classified as Vulnerable to extinction on Great Britain's new Red List for mammals due to long-term declines[12], are showing positive signs in low-density urban habitats. Since 2012, fewer urban Hedgehog sites have been lost and, where populations remain, Hedgehog numbers appear to be growing[13]. Other species appear to have shifted from their traditional habitats; Foxes and Herring Gulls, for example, increasingly occupy the UK's major cities, which can cause conflict with humans. There is concern for the conservation status of both of these species in their traditional habitats, but in urban areas where food is readily available (particularly from refuse) they are able to achieve high population densities.

Many people in urban areas feed wildlife, including birds, Foxes, Badgers, Hedgehogs, and – sometimes unintentionally – invasive non-native Grey Squirrels and Brown Rats. There is more to learn about the impacts of this supplementary food for wildlife conservation, but it is clear that many local populations benefit, and national increases in some familiar birds such as the Goldfinch have been supported by the growth in garden bird feeding[14].

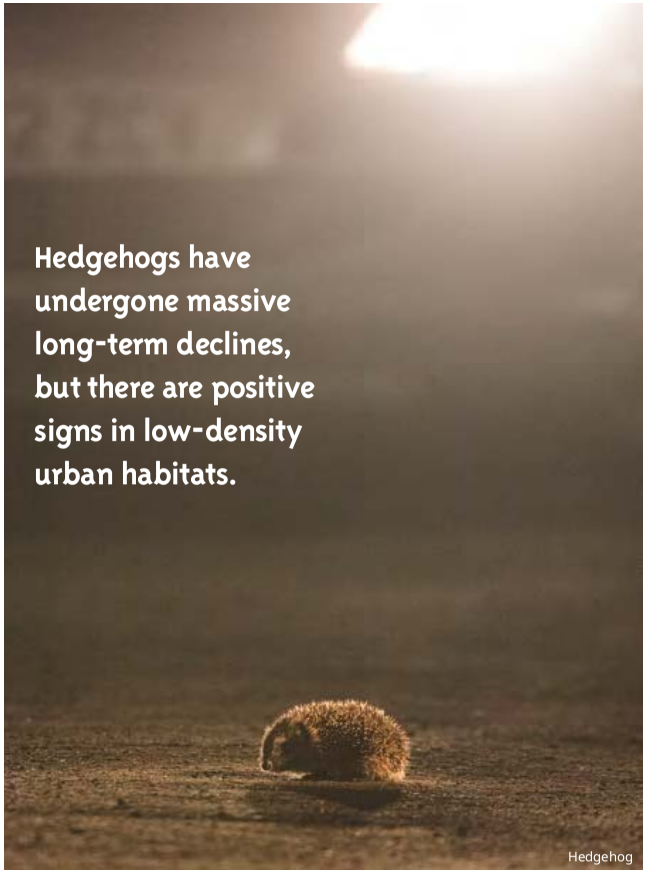


Photo: David Woodfall (rspb-images.com)

THE RESPONSE FOR NATURE

Unsurprisingly, the UK's urban areas are where most people make a connection with nature. There is therefore enormous potential for engaging people to take action in their own gardens or to act together in local green space projects. This is being supported by policy and planning regulations to ensure best practice in new developments and management of existing green spaces, as well as through empowering citizens to take individual and collective responsibility for the nature on their doorstep.

URBAN PLANNING WITH MULTIPLE AIMS

Recent changes mean that planning policy in England enshrines the concept of net gains for biodiversity through development; similar policies of net benefit were introduced in Wales in 2016. The extent to which this may be achieved is unknown, as there are no robust systems in place yet for before-and-after construction comparisons. For example, a review of a subset of development mitigation for bats in buildings found that, for a range of reasons, mitigation measures varied in their success[15]. There is increasing recognition that urban planning strategies need to be designed to provide maximum benefit for multiple aims; for biodiversity, for wider ecosystem service delivery, and the health and well-being needs of people to access urban green space. However, these aims may not always be aligned and there is a tension between intensive development, providing ecosystem services that need extensive areas (e.g. flood control), and providing accessible green space for recreational needs[16]. One solution is to employ a hierarchy of green infrastructure from larger, more distant places to smaller and more local ones.

ENGAGING WITH GARDEN WILDLIFE

Increasingly, people are being encouraged to garden with wildlife in mind, and there is the potential for substantial biodiversity benefits within the domestic gardens that constitute substantial areas of our cities.

CASE STUDIES

Connecting communities with the nature on their doorstep

Launched in March 2019 by Earthwatch Europe, Naturehood is transforming urban green spaces into wildlife havens by bringing people together.

Naturehood's approach empowers and connects people with both nature and their wider community. Individuals are supported to take action to help and monitor local wildlife in their gardens, balconies or window boxes, while Naturehood's engagement officers work across communities to create connected networks of habitat to support biodiversity.

The first four Naturehoods are now up and running in Oxford and Swindon, with support from the National Lottery Heritage Fund and in collaboration with Wiltshire Wildlife Trust.

Reducing the impact of artificial lighting on bats and insects

Artificial night lighting is one of the most pervasive – and yet under-recognised – causes of environmental pollution. Globally, external lighting is increasing by around 6% per year; in the UK we have more than 9 million street lights, and more and more buildings are illuminated at night on the grounds of security or aesthetics.

Among known impacts on wildlife, such as on night-flying insects, there is now overwhelming evidence that bat communities are profoundly affected by light pollution. Many species – including all but one of those on the Great Britain Red List – avoid lit areas for foraging or commuting, while the illumination of historical buildings (or any roosting site) at night can prevent bats accessing their roosts.

Some local planning authorities are taking steps to decrease light pollution, particularly in areas with sensitive bat species, for example by requiring forms of lighting that direct the illumination to only those areas where it is needed[17]. Individuals can help to lessen light pollution by switching off exterior lights at night, focusing security lights more precisely, choosing bulbs with lower light intensity and avoiding placing outside lights on white walls, where their effect is magnified. Simply closing curtains at night also makes an important difference!

Photo: Andy Hay (rspb-images.com)



INVASIVE NON-NATIVE SPECIES, PESTS AND PATHOGENS

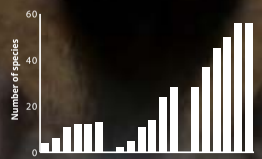
Humans are increasingly moving species around the globe and releasing them, deliberately or accidentally, into the wild. Some will be moved as contaminants of trade. The number of non-native species colonising or becoming established in the UK continues to grow. Around 12% of established non-native species cause adverse economic, environmental or societal impacts and are therefore classed as invasive non-native species (INNS). As impacts often take time to emerge, and INNS establishment and spread will be facilitated by climate change, that figure is likely to rise. While eradications of INNS from islands, for example, can have direct benefits for impacted species, preventing species from establishing in the first place is recognised globally as the best strategy for avoiding environmental damage and minimising the financial cost of subsequent management.

PRESSURE

12% of established non-native species have a demonstrated negative ecological or human impact, and that figure is likely to rise.

STATE

The number of INNS established in Great Britain has increased steadily.



RESPONSE

Simple biosecurity measures can keep INNS off islands. Breeding seabird numbers have shown a rapid positive response to successful rat eradications on UK islands, including Lundy, Canna, Ramsey, the Shiants and the Scilly Isles.

Photo: Craig Churchill (rspb-images.com)

Muntjac Deer

PRESSURES ON NATURE

The impact of non-native species has long been recognised as a major driver of environmental damage. Impacts are evident across all ecosystems, with particularly severe impacts on native woodlands, freshwater habitats and islands – which often have unique floras and faunas, due to their isolation from nearby continents. There are currently around 2,000 non-native species known to be established in Great Britain, and an average of 10–12 new species establish each year; around 12% of these cause serious adverse impacts and are therefore classed as invasive (INNS)[1,2]. Over 1,200 non-native species have been recorded in Ireland.

These figures are likely to increase as climate change proceeds. INNS can have a number of adverse impacts.

- They may outcompete or predate native species, as has happened with American Mink and Water Vole[3].
- INNS may bring with them diseases to which native species have little or no resistance, such as squirrel pox from American Grey Squirrels that is fatal to native Red Squirrels.

Photo: Paul Sawer (rspb-images.com)


Toad


Floating Pennywort

Photo: GBNNSS

- Initially harmless airborne pathogens can become more virulent as the result of hybridising with formerly benign native microbes. Planting of exotic pine stands in the UK appears to have facilitated the introduction of two exotic races of the fungus *Dothistroma septosporum* into Scotland, which now pose a threat to native Scots Pine, both directly and through potential hybridization and introgression with the endemic race[4,5].
- Hybridisation of a novel species with a closely related, native one can also risk effective extinction of the native one by loss of its unique/separate genetic identity, as is currently happening with Scottish Wildcat[6].

Novel pests and pathogens can have impacts beyond the species they directly attack. For example, the spread of Ash dieback is likely to affect the many species that rely upon Ash bark, roots or even its decomposing leaves. Declines in Ash are predicted to lead to the loss of insects, lichens, mosses and liverworts, many of which are found largely or solely on Ash[7].

Amphibian diseases have become a leading cause of the decline and extinction of many amphibian populations around the world[8]. The emerging infectious chytrid fungus *Batrachochytrium salamandrivorans* could be devastating to UK newt populations; current efforts focus on countering the risk of its introduction to the wild via amphibian collections or the international pet trade. It has already caused mass mortality events in continental Europe, leading to the near extinction of the Fire Salamander *Salamandra salamandra* in the Netherlands[9].

INNS can also reduce the benefits we gain from nature. For example, invasive plants can choke up waterways and increase flood risk, while others may cause damage to species of economic importance. In 2010 the total annual cost of INNS to the British economy was estimated at approximately £1.7 billion[10].

ON AVERAGE 10–12 NEW NON-NATIVE SPECIES NOW ESTABLISH EACH YEAR IN GREAT BRITAIN, OF WHICH AROUND 12% WILL CAUSE SERIOUS ADVERSE IMPACTS.

THE STATE OF NATURE

Globally, INNS are one of the main drivers of biodiversity loss and constitute a particular threat to fragile ecosystems, such as those on islands. Of 247 animal extinctions where the cause is known, INNS were implicated in 58%, and in 31% of cases where species have gone extinct in the wild, INNS were the only cause cited[11]. While INNS have been implicated in extinctions from UK OTs, there are none recorded from Great Britain and Ireland.

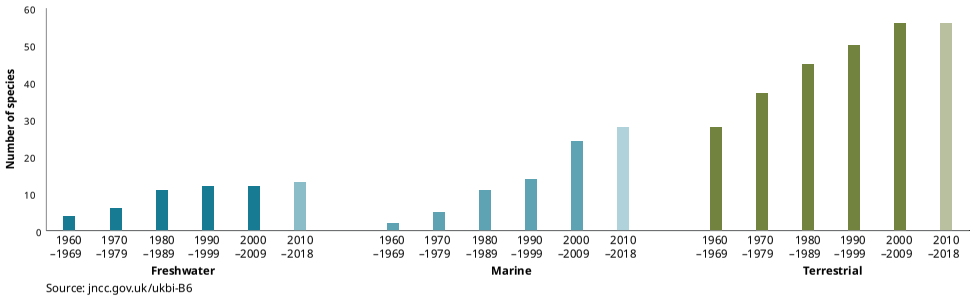
Over 3,200 non-native species have been recorded in Great Britain, with around 2,000 established and reproducing in the wild. Two hundred and thirty-four species have had a documented negative ecological or human impact[12]. The number of terrestrial invasive species becoming established in Great Britain has grown steadily and numbers of marine species have increased even more rapidly, having more than doubled since 1999 (see figure below). The greatest numbers of INNS are found in woodlands and urban ecosystems, followed by coastal, freshwater and grassland habitats[13].

OVER 3,200 NON-NATIVE SPECIES HAVE BEEN RECORDED IN GREAT BRITAIN.

RELATED SECTION

p80 UK Overseas Territories

UK Biodiversity Indicator: Number of new invasive non-native species established in or along 10% or more of Great Britain's land area or coastline, 1960 to 2018 (final columns in each habitat refer to nine years rather than 10)



Source: jncc.gov.uk/ukbi-B6

Photo: John W Anderson



THE RESPONSE FOR NATURE

While the control of established INNS is important, the GB INNS Strategy emphasises the need for prevention, early detection and rapid response to potential establishment. Stopping problems before they start makes huge savings both in terms of money and ecological damage. The EU and the UK Government have response plans to deal with new threats, and to date these have proved effective at preventing colonisation of some new species.

CASE STUDIES

Removing INNS

Although islands are often seen as being particularly vulnerable to INNS, they have the advantage of offering a contained system from which INNS can be removed. Great Britain and Ireland are archipelagos, and several successful projects to remove non-native species have been carried out here, such as the eradication of rats on the islands of Canna, Lundy, Ynys Seiriol (Puffin Island) and the Shiants. On the British mainland, projects have successfully eradicated Coypu, Muskrat, African Clawed Toad, Fathead Minnow and Black Bullhead. Eradication projects are currently underway on UK OTs, including Gough Island, Bermuda and Gibraltar.

After clearing an INNS from an area, the recovery of the impacted species or habitat can be remarkably quick. South Georgia was declared rat-free in 2018, following the largest eradication of its kind. Before baiting began in 2010, numbers of Wilson's Petrel were in the low tens, but five years later hundreds were regularly seen flying over suitable breeding habitat that had been infested with rats for over a century[14]. Such recovery is not true in all cases, however. For example, even 30 years after Rhododendron removal from Scottish Atlantic Oak woodland, the ground flora had not returned to its target condition[15].



Photo: Kate Lawrence (rspb-images.com)



Photo: GBNNSS

INNS awareness

Action plans include ways to reduce the risk of zoo escapes, contaminants from angling equipment and introductions via recreational boating. Invasive freshwater plants and animals can spread quickly and impact whole ecosystems. Killer Shrimp, widely considered one of the most damaging INNS in Europe, spread from the Ponto-Caspian region of Eastern Europe. There are three established populations in the UK, elsewhere records are mainly from eastern and central Europe to date. Killer Shrimp can cause population declines of many native species, preying on other shrimp species, fish larvae and eggs[16].

Campaigns such as Check, Clean, Dry and Be Plant Wise[17] aim to improve biosecurity by raising awareness with key stakeholders, anglers and boat users, in order to slow the spread of such invasive freshwater species and prevent their establishment in new areas. However, the latest analyses suggests that public awareness of INNS issues has not increased over the past decade, although awareness has improved among key groups such as anglers and boaters[18].

POLLUTION

A wide range of pollutants, from many sources, threaten wildlife and have an impact on all habitats. Perhaps the most widespread current harm is caused by excess nutrients (phosphate and compounds of nitrogen) in air and water. Legislative control has been successful in reducing emissions of many pollutants and there are signs of a partial recovery in nature, including fish returning to rivers, and lichen and bryophyte species (that are sensitive to air pollution) expanding in range. Sensitive habitats in particular, however, remain vulnerable. New government initiatives, partly in response to international targets, should lead to further improvement. Public concern regarding pollution has been strengthening recently, particularly over the proliferation of plastic waste in the environment, and there is a widespread appetite for positive change.

PRESSURE

Nitrogen and ammonia emissions have decreased since 1970.



STATE

Occupancy trends in lichens and bryophytes suggest some recovery.



RESPONSE

The UK has committed to legally binding limits for sulphur dioxide, reactive nitrogen, ammonia and particulate matter by 2020, and further reductions by 2030.

Photo: Chris O'Reilly (rspb-images.com)

PRESSURES ON NATURE

There are few places left on earth unaffected by the by-products of modern human life. Pollution presents a wide range of threats to the environment and the species that inhabit it, as well as to human health and well-being. Pollutants come in a diverse range of forms, including but not limited to plastic waste; chemicals in water, soil and air; noise and light emitted from cities and transport; and nutrient enrichment of sensitive habitats.

Air pollution and nutrient enrichment affect biodiversity and ecosystem services, harm human health and contribute to climate change. Widespread changes have been recorded to sensitive ecosystems in the UK, with farming, transport and industry being key pollution sources. Eutrophication, acidification and toxic pollution of ecosystems have been shown to drive declines in the presence, abundance and health of sensitive species of plants, lichens and other fungi[1]. There is strong evidence that nutrient enrichment via nitrogen deposition has impacted plant species in a wide range of habitats[2].

Diffuse pollution represents a significant risk to freshwater in the UK, and arises from the run-off of soil, nutrients and pesticides from farming and forestry, contaminated drainage from urban areas and the deposition of pollutants from the air. Point source emissions from industry, sewage treatment works and fish farming add to this pollutant load. High levels of phosphates and nitrates are the most frequent causes of rivers, lakes, estuarine and coastal waters failing EU Water Framework Directive (WFD) water quality objectives[3,4], and impacts can linger for years.

Recent decades have seen reductions in many forms of pollution, most notably point source discharges from industry and water treatment works, but others remain difficult to combat. New threats have arisen with the introduction of novel agrochemical and pharmaceutical products[5]. Emissions of reactive nitrogen (NOx), sulphur and particulates into the atmosphere have decreased since 1970, though NOx remains a key source of nitrogenous pollution to the atmosphere. Ammonia emissions (which contribute to the formation of NOx) fell overall by 11% between 1980 and 2017, but have risen since 2013 and constitute a major short-term threat to sensitive habitats. Dairy and beef cattle, fertiliser application and poultry are major sources of ammonia emissions[6].

Trends in annual emissions of various air pollutants, 1970 to 2017[6]

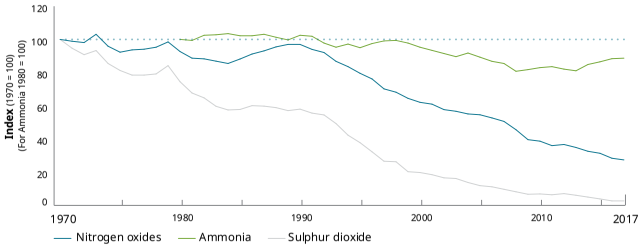


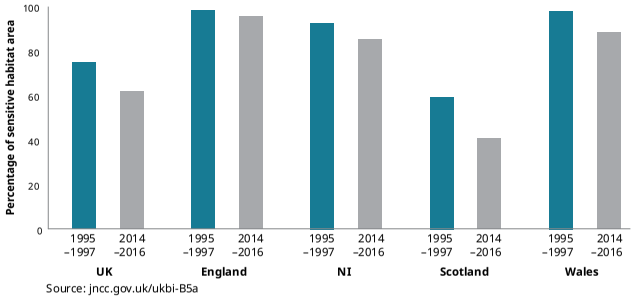


Photo: Ray Kennedy (rspb-images.com)

THE STATE OF NATURE

Most semi-natural habitats, and over two-thirds of our wildflowers (such as Harebell and Betony), require levels of nitrogen to be low[7]. Nitrogen enrichment has led to changes in plant and fungal communities and the declines of individual species, with species that prefer nutrient-poor environments showing a decline in distribution, on average (see graph below). Further up the food chain, moth species whose larvae depend on low nutrient-adapted plants declined strongly between 1970 and 2010[8]. Despite recent progress in input reduction, in 2014–16 62% of all UK sensitive habitats exceeded the recommended "critical nutrient load", above which they are at risk of harmful effects. The figure is 85% for Northern Ireland and for England is close to 100%[9].

UK Biodiversity Indicator: Percentage of nitrogen-sensitive habitat area in UK countries where nutrient nitrogen critical loads were exceeded in 1995–97 and 2014–16

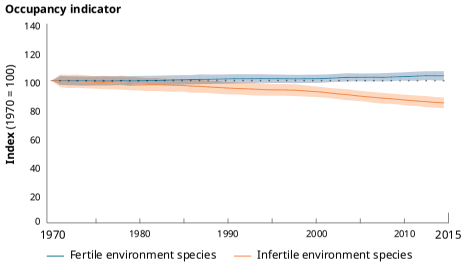


Source: jncc.gov.uk/ukbi-B5a

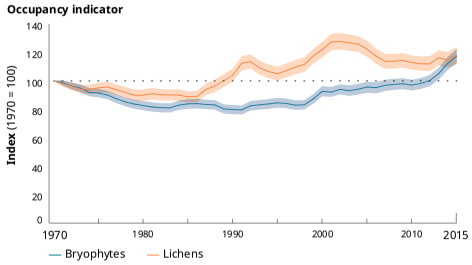
Many waterways are now cleaner than they have been for many years. In 1957, the Natural History Museum declared the lower Thames "biologically dead", with news reports describing it as a vast, foul-smelling drain. Now it is used by a wide variety of fish, with Atlantic Salmon, Brown Trout and River Lamprey all recorded in recent years[10]. However, despite these improvements, only 35% of UK surface water bodies were classed as being in high or good status (based on the WFD combination of biological, chemical and structural measures) in 2018[11], and species such as the Freshwater Pearl Mussel continue to suffer declines related to water quality.

New trends shown here (below) reveal interesting patterns in the distribution of bryophyte and lichen species since 1970. While the drivers of this recovery are not fully understood, the increases in occupancy began in the 1990s, as levels of air pollution started to reduce, and is almost certainly related to the recovery of species sensitive to pollution[12]. This is likely to reflect an increase in the distribution of common species sensitive to acid deposition but tolerant of increasing nitrogen deposition, reflecting changes in air pollution. Evidence suggests, however, that many specialist lichen species continue to decline in abundance as well as distribution[13].

Change in the distribution of species of fertile environments (411 species) and infertile, nutrient poor, environments (524 species) based on Ellenberg N scores



Change in the distribution for bryophytes (569 species) and lichens (696 species) 1970 to 2015



THE RESPONSE FOR NATURE

The marked reductions in many classes of polluting emissions has been driven by broad-scale legislative control, often following coordinated action across the UK and the rest of Europe[14]. Sulphur emissions from coal-fired power stations and industry in the UK, which caused "acid rain" and damaged ecosystems across northern Europe, have been cut by 97% since the 1970s[6]. Further reductions in sulphur emissions are expected when a mandatory 0.5% cap of sulphur content in shipping fuel comes into force on 1 January 2020[15].

The reduction of NOx has been part of government clean air strategies for many years, driven largely by concerns about public health impacts and greenhouse gas emissions. Recently, measures to reduce ammonia emissions from agriculture and other sources have been adopted or are under consideration across all UK countries (for example, the Department for Environment, Food and Rural Affairs (Defra)'s Clean Air Strategy 2019), driven by binding international targets to reduce emissions by 8% by 2020 and 16% by 2030 from a 2005 baseline[16]. All current UK AES now contain options or mandatory cross-compliance obligations to protect watercourses from farming-related diffuse pollution and Nitrate Vulnerable Zone regulations impose additional protection measures in many areas (for example, they cover around 55% of English agricultural land).

CASE STUDIES

Catchment Sensitive Farming

Since 2006, the Catchment Sensitive Farming (CSF) partnership has worked with over 20,000 farmers, covering a third of the farmed area of England to improve water quality. Workshops, farm events and one-to-one advice are targeted in areas designated as high priority for agricultural pollution. Topics covered by the programme include fertilizer and pesticide management; soil health and structure; silage, slurry and fuel oil regulations; and Nitrate Vulnerable Zones. Field officers also provide help and advice with grant applications for farm infrastructure upgrades and capital works. In target areas monitored since Phase 1, nutrient, sediment and faecal indicator organism concentrations have reduced by 5–22% and monitored pesticide concentrations (exceeding 0.1µg/l) by 34%[17].

A key element of the programme aims to develop local farmer–to-farmer learning networks. One example in the Evenlode catchment, Oxfordshire, involves farmers alongside the local community measuring the effectiveness of CSF interventions through FreshWater Watch, Earthwatch's citizen science project. The data collected encourage better catchment management and empower the rural community to act as stewards of the river.

Moninea Bog, Northern Ireland

Moninea Bog, an Area of Special Scientific Interest (ASSI) and Special Area of Conservation (SAC), is a lowland raised bog in the south west of Northern Ireland, which is especially important for its high cover of Sphagnum moss species and the presence of all three native sundew species. In 2007, a field study by the Centre for Ecology & Hydrology showed that ammonia from neighbouring farm operations was causing severe direct damage to the site's vegetation, including algal slime on trees and bleaching of mosses. Following a reduction in the level of farm operations in 2009/2010, further investigation in 2017 found that there was already substantial recovery on the site. The results are now being prepared for publication, but already provide an example that measures to reduce ammonia emissions can be expected to deliver substantial biodiversity benefits within five to 10 years.

BROAD-SCALE LEGISLATIVE CONTROLS HAVE DRIVEN DOWN EMISSIONS OF MANY POLLUTANTS.

Photo: David Genney



WOODLAND MANAGEMENT

The area covered by woodland in the UK continues to increase from very low levels a century ago, but its integrity is under threat from invasive plants, pests and diseases. Nature in woodland is under pressure from a lack of management, overgrazing by deer, increasing levels of recreational disturbance and nitrogen pollution. Numbers of many woodland birds and butterflies continue to decline. Of particular importance due to their high nature value and restricted range, the Atlantic oak and hazel woods of the UK's western coasts and Scotland's Caledonian pine forests have both suffered severe loss and fragmentation. Ancient woodlands across the UK have been lost through conversion to plantation forestry and face continued threat from infrastructure and housing development. Addressing the problems facing woodland and trees of the wider countryside is increasingly recognised as a major conservation issue, which is the focus of a wide range of ambitious consortium projects involving both research and conservation action.

PRESSURE

Woodland area in the UK has increased since 1998.



STATE

The UK woodland butterfly indicator has fallen by 50% since 1990.



RESPONSE

Area of woodland sustainably managed, about 44% of the total.



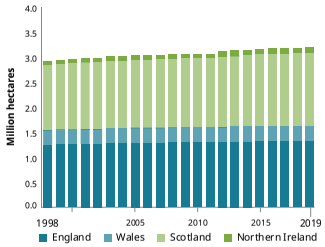
Photo: David J Slater (rspb-images.com)

PRESSURES ON NATURE

Looking back in history woodland cover in the UK declined gradually, largely to make way for agriculture. Estimated at 15% in England at the time of the Domesday survey, it had fallen to 5% by the end of the First World War. In 1919, the Forestry Commission was established to provide a strategic timber reserve and, assisted by policy incentives to encourage private planting, the area of woodland increased rapidly. Although well below the European average of 37%[1], woodlands now cover around 13% of UK land. However, non-native conifers such as Sitka Spruce dominated early restocking, and with limited tree species, age diversity and high-density planting, this benefited only a narrow range of wildlife, and eliminated species that depended on habitats on which these woods were planted on. With growing recognition of the conservation value of native woodland, many new woods and replantings now use diverse tree mixes or allow natural regeneration and follow sustainable management practices, in accordance with the UK Forestry Standard.

UK woodland cover increased by 9% between 1998 and 2018 and is currently estimated at 3.17 million ha. Scotland has seen the largest area increase (156,000ha), while Northern Ireland had the greatest proportional increase (39%) but remains the least wooded UK country. Conifers account for 51% of the UK's woodland area, varying from 26% in England to 74% in Scotland[2]; much of this is commercial plantation of non-native tree species. Ancient woodland, highly important in terms of biodiversity value and supporting a wide range of specialist species is estimated to cover only around 2.4% of UK land and the lack of an up-to-date inventory hampers its protection.

Woodland area by country, 1998 to 2019



The pressures affecting woodland, and trees in the wider countryside, are diverse and dependent on location and species. Changes in management have had major effects. Historic rotational coppice systems were replaced by clear fell and replanting during the 20th century, then followed by a recent move to continuous cover systems, often with very limited management. Many woods have become fragmented, intersected by roads and development that degrade habitat and form barriers to wildlife movement. Increasingly, the prevalence of tree disease is a serious concern. Dutch elm disease resulted in the loss of 20 million trees during the 1970s[3] while Ash dieback and Acute Oak Decline are currently seriously affecting three of our most common and widespread tree species. The pathogen *Phytophthora ramorum* mainly affects Larch within plantations but is known to infect other tree and shrub species; its arrival has led to widescale preventative felling.

Increasing deer numbers (both native species such as Roe and non-natives such as Muntjac), have a heightened impact on woodland and its dependent wildlife as they reduce natural regeneration and alter woodland structure through increased grazing and browsing[4]. Recreational use, particularly in woodland close to urban areas, has detrimental impacts on soils, invertebrates and flora through trampling and compression[5]. Disturbance associated with an increasing variety of leisure activities adversely affects wildlife, including birds and mammals[6]. A changing climate in combination with invasive and native pests and pathogens form a complex and interacting set of threats that will favour some tree species, while making conditions worse for others. Species' distributions are expected to alter as a result, with evidence already pointing to a decline in the abundance and distribution of woodland specialist lichens[7].

Population trend for Muntjac deer – UK Breeding Bird Survey Index, 1995 to 2018

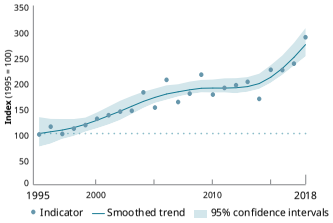




Photo: Norman Russell (rspb-images.com)

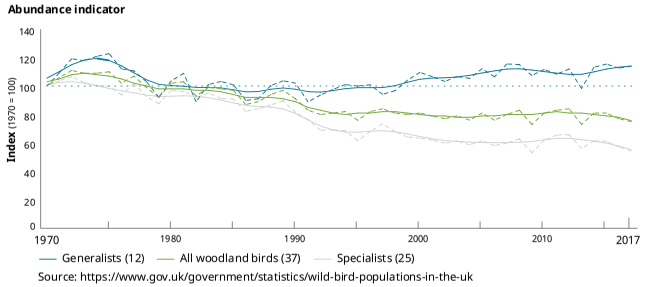
THE STATE OF NATURE

Despite the overall expansion in the area covered by woodland, woodland species continue to decline, as evidenced by the UK woodland bird indicator which fell by 25% between 1970 and 2017.

Declines are most pronounced in woodland specialists such as Lesser Spotted Woodpecker, Spotted Flycatcher and Willow Tit, and suggested causes include a decline in appropriate woodland management, changes in tree species composition and loss of landscape connectivity. Woodlands acquire species at different stages of maturity, with specialist bird species generally occupying woods over 80 years old, and with a high level of structural diversity, while generalist species can exploit younger woodland[8].

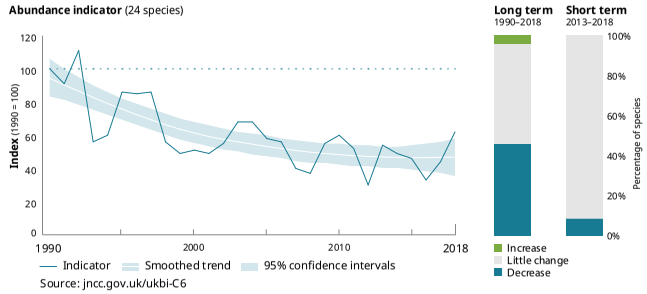
The long-term decline in the UK woodland butterfly indicator, which includes well-known species such as Common Blue, Marbled White and Meadow Brown, is thought to be due to reduced sympathetic management and a decline in open spaces within woods.

Breeding woodland birds in the UK, 1970 to 2017



Source: https://www.gov.uk/government/statistics/wild-bird-populations-in-the-uk

UK Biodiversity Indicator: Change in widespread butterflies in woodland in the UK, 1990 to 2018



Source: jncc.gov.uk/ukbi-C6

Photo: Colin Wilkinson (rspb-images.com)



THE RESPONSE FOR NATURE

Grants are available to encourage new woodland planting and the sustainable management of existing woodland. For example, funding is available under Rural Development Programmes to deliver local biodiversity objectives (for priority habitats and priority species), water objectives (to improve water quality or help reduce flood risk) and climate change mitigation or adaptation[9,10].

As the figure below shows, the area of woodland managed sustainably (which includes measures relating to maintaining biodiversity, ecological, economic and social functions) has remained level, at around 43% (44% in 2019), over the last decade. The Scottish Government has a stated aim to increase woodland cover from around 18% to 21% of Scottish land area by 2030[11] although much of this will be non-native planting, limiting its biodiversity value. In Wales the latest Government strategy states an ambition to create 100,000ha of new, well-located woodland by 2030 to help Wales meet its carbon emission reduction targets[12]. Planting targets can deliver the greatest benefits if they ensure new woodland is in the right place and not compromising other valuable habitats; priority is given to creating both new native and new mixed woodlands that can deliver multiple benefits.

UK Biodiversity Indicator: UK area of forestry land certified as sustainably managed, 2001 to 2019



Source: jncc.gov.uk/ukbi-B1b

The importance of maintaining, increasing and enhancing woodland for nature conservation alongside a range of ecosystem services is widely recognised in a number of ambitious consortium projects across the UK. Cairngorms Connect aims to enhance habitats, species and ecological processes over 600km^2 of Scotland's Cairngorms National Park, including Caledonian pinewoods and montane willow communities[13]. In England, the Woodland Trust is leading a consortium project to plant 50 million new trees over 25 years to create a "Northern Forest", spanning the country from Hull to Liverpool[14]. Launched in 2019, the online Woodland Wildlife Toolkit[15] provides woodland managers with a one-stop resource to inform about distributions of important wildlife, assess woodland condition and create appropriate management plans.

CASE STUDIES



Photo: Mike Read (rspb-images.com)

Celtic Rainforest Project

The Celtic rainforests – Atlantic woodlands in Ireland and the UK with an open woodland structure, mild and humid conditions, and rich plant assemblages – comprise a large proportion of Europe's rare temperate rainforest habitat. In the UK they are threatened by invasive non-native plants (particularly Rhododendron), poor grazing regimes, a lack of management and atmospheric nitrogen deposition[16]. By tackling invasive plants, developing restoration techniques and demonstrating best practice, the Celtic Rainforest Project aims to increase resilience, enhance ecosystem function and return woods to a favourable conservation status across four areas of north and mid-Wales. This ambitious EU LIFE-funded project runs from 2018 to 2025 and involves a range of statutory and NGO partners, led by Snowdonia National Park Authority.



Photo: Richard Revels (rspb-images.com)

Ash dieback

Ash dieback (caused by the fungus *Hymenoscyphus fraxineus*) is one of several invasive tree diseases recorded in the UK recently that has the potential to cause major ecological, social and economic impacts. With an estimated 126 million trees within woodland and 60 million in the wider countryside, Ash is the UK's third most common tree[17]. First recorded on imported saplings during 2012, the disease has spread rapidly to all four UK countries. Recent analysis has shown 955 species, ranging from bryophytes to mammals, are associated with Ash, of which 45 are obligate (only occurring on Ash) and a further 62 are highly associated[18,19].

The disease is now well established, and the economic cost has been estimated to be in the order of £15 billion[20]. There are many uncertainties over the prognosis for native Ash woodlands; mature trees die slowly and most populations show some level of tolerance; this has been shown to be strongly heritable. While a worst-case scenario would see loss of a high proportion of Ash, lessons from other epidemic tree diseases suggest that maintaining a wide genetic base and reducing other stressors will maximise tolerance. Guidance[20] focuses on ameliorating the potential impacts and encouraging ecological functioning by retaining existing Ash trees, encouraging natural regeneration of Ash and other native species, reducing herbivore impacts and reducing competition from INNS. Research has sought to establish which other tree species are able to support Ash-associated species and several, including Oak, Elm and Beech (where native), can support at least half[21].

WORKING FOR NATURE: CONSERVATION IN THE UK

The preceding pages have reviewed the biggest drivers of change acting on the UK's nature, how these pressures have changed over recent years and their impact on wildlife, both in the past and now. We have also given examples of some of the inspiring conservation action taken in response to these pressures. Here we look at how conservation acts as a force to help nature, to mitigate against the worst human impacts on nature and to nurture recovery from decades or even centuries of pressure.

Photo: Malcolm Hunt (rspb-images.com)

An analysis of drivers of change, reported in *State of Nature 2016*, estimated that 19% of the total impact (17% positive, 2% negative) on nature in recent decades arose from a wide array of conservation activities such as nature-friendly farming, habitat management and the creation of new wildlife habitat. This demonstrated that the conservation response can work, and is backed up by a number of well-evidenced examples of conservation success (see conservationevidence.com). Harder to assess is the extent to which the state of nature would be worse without this conservation action. We know, for example, that there are continued declines in biodiversity in response to farmland management (see page 20) – but we also know that nature-friendly farming, supported by government-funded AES, can deliver a range of on-the-ground benefits for wildlife. The key, it appears, is a matter of scale and resources. The collective response to the pressures on nature is helping, and should be celebrated, but declines in nature are continuing.

Here, we give a brief overview of just some of the approaches used in the conservation of the UK's wildlife, and the challenges we face in trying to improve the state of nature.

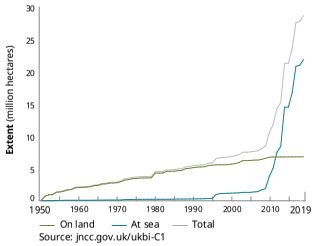


Photo: Colin Wilkinson (rspb-images.com)

CONSERVING SPECIAL PLACES

The UK has a long heritage of preserving special places, with the world's first nature reserve declared in West Yorkshire in 1821. As pressures on the wider environment have increased, with the widespread loss of habitats such as heathland, wetlands and ancient woodland, such reserves have become ever-more important. The State of Nature partnership is collectively responsible for approaching 5,000 individual nature reserves, with a combined area over five times the size of Greater London. In addition, the designation of sites and landscapes of high nature value as ASSIs or Sites of Special Scientific Interest (SSSIs), Special Areas of Conservation (SACs) and SPAs provides special legal protection to prevent damaging activities and thus conserve important habitats and species. There has been a massive increase in the area of designated land and sea in the UK, owing to the designation of inshore and offshore Marine Protected Areas (MPAs) since 2013, although designation, of itself, does not ensure effective conservation.

UK Biodiversity Indicator: Extent of UK nationally and internationally important protected areas, 1950 to 2019



Source: jncc.gov.uk/ukbi-C1

Protection is only part of the story: nature reserves and protected areas require careful and nuanced management to enable species and communities to flourish. In 2019, the proportion of protected sites assessed as being in favourable condition was 43% for SACs, 50% for SSSIs and 52% for SPAs, with the majority of non-favourable sites assessed as "unfavourable-recovering". Very few habitats are stable and most require management of grazing levels, woodlands or hydrology, and in some cases management must evolve in response to an improved understanding of the habitat or species' requirements. At the same time, reserves must be able to welcome in people, and thus balance the needs of both visitors and the wildlife they come to experience.

CASE STUDY

Redgrave and Lopham Fen in Suffolk is the largest remaining calcareous valley fen in England, but decades of water abstraction and nutrient enrichment caused huge degradation to the site. Restoration work, funded by the EU and led by the Suffolk Wildlife Trust, relocated the abstracting borehole, removed 77ha of encroaching scrub, and scraped 23ha of fen completely clear to enable recovery. Forty-two ponds were created to help the threatened Fen Raft Spider. The rewetted fen holds 270 plant species and a rich invertebrate community; extinction of the Fen Raft Spider has been avoided and it is now expanding successfully in the new habitat.

RESTORING LANDSCAPES

The need to think big to turn around the fortunes of nature has been increasingly well recognised; such thinking that was crystallised in Professor Sir John Lawton's 2010 report, *Making space for nature*[1], which called for "more, bigger, better, joined" wildlife sites. Vitally, this extends conservation thinking beyond reserves, to the need to reduce fragmentation – whereby nature is stuck in isolated islands in a hostile sea of intensively managed countryside – by creating corridors between sites for wildlife to flow through, or stepping stones to help species jump across. In addition, Lawton called for "buffers" around wildlife sites, softening the intensive management that often runs up to reserve boundaries. Finally, Lawton's call for "better" needs to be addressed. The management of large areas of potentially wildlife-rich semi-natural habitat, such as in the UK's uplands, is unsympathetic to biodiversity.



Photo: David Wootton (rspb-images.com)

CASE STUDY

Great strides are being made to introduce such approaches, and demonstrate partnership working between government, NGOs, businesses, farmers and local communities. For example, in Northern Ireland, NGOs including the RSPB, Ulster Wildlife and Butterfly Conservation are working with local and national government to protect, restore, expand and link wet grassland habitats across 2,000km^2 of the Lough Erne Basin, home to threatened wader populations and rare plants such as the Fen Violet and Irish Lady's-tresses Orchid.

FOCUSING ON SPECIES

While conservation policies to address the pressures on nature, delivered across landscapes, will help, for some species this will not be sufficient. Many of the most celebrated conservation successes of recent years – the return of the Pine Marten to Wales, the restoration of the breeding range of Red Kites, the establishment of Lady's Slipper Orchid at 11 sites in Northern England – have been the consequence of targeted action, based on robust science and conservation best practice. Programmes such as Back from the Brink, in England, have brought numerous partners together to target action for priority species, including the Lesser Butterfly Orchid, Barberry Carpet Moth and Ladybird Spider.

A review of the 1,063 terrestrial and freshwater species listed as priorities on the UK Biodiversity Action Plan (BAP) – which has now been replaced by conservation policies at the devolved country level – found that only 114 of these have been the focus of coordinated and targeted action in at least part of their UK range as part of species recovery projects (SRPs). Most of these had received little or no conservation action aimed explicitly at recovering their populations[2]. However, there are imbalances in the targeting of such efforts. While conservation action targeted at one species can help others – for example, reedbed creation for Bitterns is likely to have benefited a wide range of taxa, including Water Voles and the Reed Leopard moth[3], and agri-environment options intended to boost rare farmland birds such as Stone-curlew have benefits for a range of taxa, including threatened arable plants[4] – our invertebrates and plants are clearly receiving less specific attention than mammals and birds. This contrasts with growing evidence that insects are showing rates of decline that may be greater than other taxonomic groups. Of those species that have been the subjects of SRPs, 61% are vertebrates, despite this group making up just 9.5% of the terrestrial and freshwater species identified as priorities. A further 26% of species with SRPs are invertebrates (mainly butterflies), despite this group representing 39% of priority species, and only 13% of plant and fungi species were identified as having had SRPs, despite this group making up 52% of all species listed as UK BAP priority species[2].



Steve Knell (rspb-images.com)

BROADER ENVIRONMENTAL POLICY

Our impact on the UK's land and seascapes is pervasive – the impact of climate change is felt everywhere, three-quarters of land is affected directly by farming policies, 33% of quota managed fish stocks are harvested unsustainably, urbanisation is widespread. So, clearly, we need policies that ensure that the need for space to live, work and play, food production, and use of other natural resources is met sustainably, in a way that allows nature to flourish. We have talked, for example, about how policies to encourage wildlife-friendly farming can be integrated with food production and the needs of nature (page 21), and how legislation to control acidifying pollution has enabled the recovery of some bryophytes and lichens (page 41). At sea, policies that prevent the overexploitation of vulnerable fish stocks, encourage fishing techniques that minimise bycatch and habitat damage, and protect the most important areas from fishing are vital to ensure healthy marine ecosystems (page 63).



Photo: Colin Wilkinson (rspb-images.com)

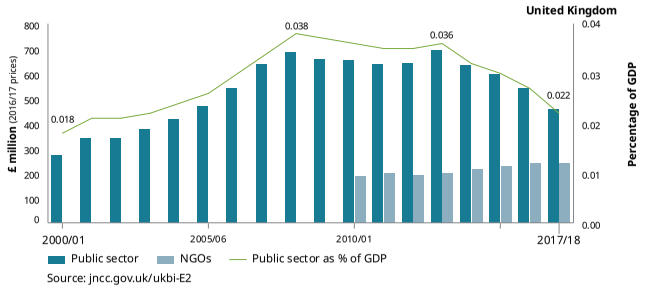
It is well recognised that policies to protect biodiversity and the environment bring huge benefits to human well-being, from clean air and water, healthy soils for food production, and the health and well-being impacts that result from connection with nature. Globally, these issues are recognised by the UN's Sustainable Development Goals[5] and by the CBD. We discuss the UK's progress towards the global 2020 targets (Aichi targets) on pages 90–91. As the parties to the CBD begin to discuss a post-2020 framework, we wait to see whether this will contain policies, and associated goals, that will encourage the transformative change which the recent Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) report stated was required to avoid global declines in nature from continuing.

RESOURCES FOR CONSERVATION

Ultimately, our ability to act to conserve the UK's nature is constrained by resources. In 2017/18, an estimated £456 million of UK public sector funding was spent on biodiversity in the UK. This funding has been declining, by 29% over the last five years, and by 34% since a high point in 2008/09. As a proportion of GDP this represents a fall of 42% from 0.038% to 0.022%. It should be noted, however, that the lower level of public sector funding for international biodiversity conservation (£205 million in 2017/18), including that in the UK's OTs, has risen by 111% over the last five years.

By contrast, spending on biodiversity in the UK by NGOs with a focus on biodiversity and/or nature conservation, while not matching government investment, has increased in recent years. It reached £239 million in 2017/18, having increased by 24% over the previous five years.

UK Biodiversity Indicator: Expenditure on biodiversity in the UK, 2000/01 to 2016/17



Source: jncc.gov.uk/ukbi-E2

The increase in NGO spending on conservation provides evidence for increased public concern for the state of nature, and the value which they place on it, as does the rapid increase in volunteering to help nature conservation (page 10). Volunteers donate an immense resource to conservation in the UK; for example, we have estimated that around 7,500,000 volunteer hours go into collecting the biodiversity monitoring data upon which the *State of Nature* reports rely, every year. So, although financial investment is crucial, as are government policy and legislation, we must remember that the most successful conservation action arises from partnerships, across governments, charities, business, landowners and individuals working together.

MARINE KEY FINDINGS

From seabirds and marine mammals at the top of the marine food web to plankton at the bottom, there have been large spatial and temporal changes in species' abundance and distribution in UK seas in recent decades[1].

Monitoring the marine environment is a more logistically challenging task than monitoring terrestrial habitats. Species data are collected across large areas, but geographic coverage varies and long time series are only available for a more limited set of taxa. Monitoring is carried out for seabirds at breeding colonies around the UK and for marine mammals and fish stocks in the Greater North Sea and Celtic Seas.

Other aspects of marine nature are monitored in relation to specific drivers of change in the marine environment. Pressures interact with each other and, in some cases, impacts are difficult to disentangle, often having synergistic and cumulative effects. For this reason, following our key metrics of the state of marine nature, we present metrics for taxonomic groups associated with specific drivers, namely climate change and fisheries, to give an overall picture of the state of marine nature.

Photo: Paul Sawer (rspb-images.com)

The UK Marine Strategy[1] provides a framework for protecting the UK marine environment while allowing sustainable use of marine resources. The update in 2019 provides a status assessment of marine biodiversity and drivers of change, closely related to broader geographic assessments across the North East Atlantic[2,3]. We draw on these assessments in the *State of Nature 2019* marine section.

MARINE FISH – CELTIC SEAS AND GREATER NORTH SEA

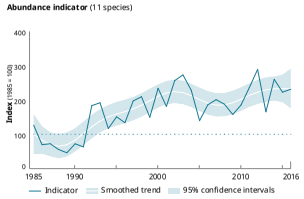
The abundance of marine fish and the composition of marine communities have been influenced by commercial fishing and climate change. Over the long-term period for which data are available (1980s to 2017), increases are evident as a result of warming sea temperatures which have enabled a large proportion of smaller-bodied pelagic fish species (e.g. Sardine and Sprat) to increase in abundance[4]. Through the 1980s and 90s fishing pressure led to declines in a number of larger-bodied species such as Cod[5] (page 59). Over the last 10 years, however, improved fisheries management has allowed some commercially fished species to increase from very low baselines.

The abundance indicators below use data from two trawl surveys[6,7] for a small proportion of the hundreds of demersal fish species that live on or near the sea floor (e.g. Cod, Haddock, Saithe). Very little is known about the vast majority of unmonitored and unregulated fish populations. The Celtic Seas (based on 11 species) and the Greater North Sea (based on nine species) indicators both show increases in average abundance as follows:

Celtic Seas

- Demersal species indicator shows a statistically significant increase of 133% over the long term (1985–2016) and a non-significant increase of 22% over the last 10 years.
- Over the long term 27% of species showed strong or moderate decreases and 64% showed strong or moderate increases; 9% showed little change.

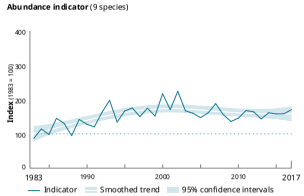
Change in average species' abundance – Celtic Seas demersal fish



Greater North Sea

- Demersal species indicator shows a statistically significant increase of 58% over the long term (1983–2017) and non-significant trend of -5% over the last 10 years.
- Over the long term 11% of species showed strong or moderate decreases and 78% showed strong or moderate increases; 11% showed little change.

Change in average species' abundance – Greater North Sea demersal fish



Data for pelagic fish species that live in the water column (e.g. Herring, Blue Whiting and Mackerel) indicate increases in average abundance in both the Celtic and the Greater North Seas over the same period. Groundfish surveys are less reliable for schooling pelagic species and therefore the direction of trend is more appropriate to report than the average magnitude of change.

Photo: Hans-Petter Fjeld


Cod

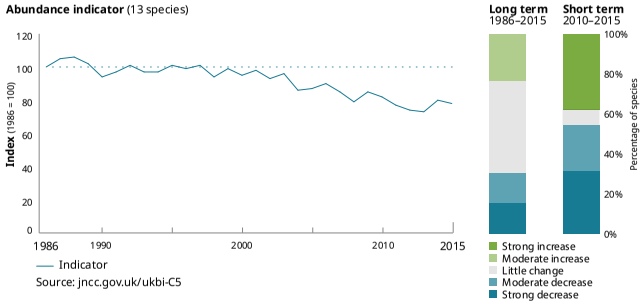
BREEDING SEABIRDS

The last comprehensive census was in 1998–2002 and reported over 8 million seabirds breeding in Great Britain and Ireland annually[8]. The latest census, Seabirds Count, is underway, and will conduct a full survey of nearly 12,000 known breeding colonies. It will provide a comprehensive update on the status of all 25 seabird species breeding in the UK. In order to dedicate time to the census, updates of the annual Seabird Monitoring Programme report have been put on hold for two years, but the most recent update of the Seabird Indicator is presented here.

Since 1986, the Breeding Seabird Indicator for 13 species declined by 22% and by 6% between 2009 to 2014.

Between 1986 and 2015, 31% of species showed strong or moderate decreases in abundance and 23% showed moderate increases; 46% showed little change. Over the short term, between 2009 and 2014, 54% of species showed strong or moderate decreases and 38% showed strong increases; 8% showed little change.

UK Biodiversity Indicator: Breeding seabirds in the UK



Regional differences are apparent for seabird populations, particularly those linked to the impact of climate change and fisheries, and declines appear stronger in the Greater North Sea than the Celtic Seas and the English Channel[9,10,11,12].

MARINE MAMMALS

Marine mammals, particularly cetaceans, are challenging to monitor, but our knowledge of their abundance and distribution is growing. Since 1994, there have been three large-scale surveys to estimate cetacean abundance in European waters, the most recent in 2016 yielding abundance estimates for nine cetacean species that regularly occur in UK waters. There is, however, only sufficient data to calculate trends for three species in the North Sea (Harbour Porpoise, White-beaked Dolphin and Minke Whale). While the areas surveyed are not wholly consistent between surveys and confidence in trends is low, numbers of all three species appear to have remained stable[13,14]. For the other six species for which trends are not available, abundance estimates were similar to or larger than previous estimates for comparable areas.

Seals are easier to survey as they haul out on land to moult (Harbour Seal) or breed (Grey Seal) and consistent monitoring has been carried out regularly for both species around UK coasts since at least the 1990s and 1980s respectively[15]. Trends have been assessed between 1994 and 2014 for the UK Marine Strategy update[16], which reports annual increases in Grey Seal numbers, but trends for Harbour Seals vary. Since 1994, Harbour Seal abundance has decreased in colonies on the north and east coasts of Scotland but increased on the east coast of England and in west Scotland (an area which holds over 20% of UK Harbour Seals). Elsewhere in the Celtic Seas there is insufficient data to assess status.

The wide-ranging nature of these marine mammals means that influences outside of the survey area may affect abundance and distribution. Marked distributional changes have been recorded in Harbour Porpoise; previously found primarily in the northern parts of the North Sea, the centre of their range has shifted southward[14,16].

Photo: Ben Andrew (rspb-images.com)



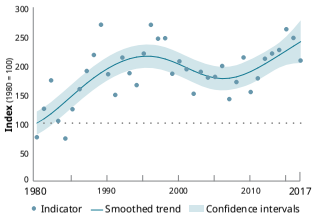
PLANKTON – THE BASE OF THE FOOD WEB

The base of the marine food web is formed by widely varied groups of organisms, including phytoplankton (photosynthetic plant-like microscopic organisms) and zooplankton (animal plankton including copepods and larvae of some commercially important species of crabs and lobsters), which are sensitive to changes in nutrients, salinity and temperatures.

Plankton communities respond rapidly to environmental changes, making them a good indicator of change in marine ecosystems although attributing observed change to specific drivers can be difficult. Plankton communities are undergoing directional change and trends in plankton lifeforms (organisms with the same functional traits, meaning that they occupy a similar niche or feed in a similar way) imply change in plankton community functioning[17].

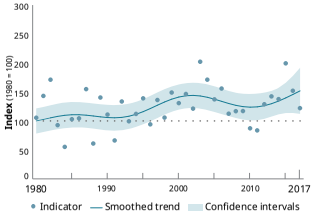
Changes in phytoplankton and zooplankton abundance vary by sea area and by group. Data are available to calculate an indicator of phytoplankton biomass, using a proxy measurement of ocean colour from the Phytoplankton Colour Index (PCI)[18] and relative abundance indicators for phytoplankton (total diatoms and dinoflagellates) and for zooplankton (small and large copepods) since 1958. Graphs on this page show examples of this variability, with contrasting patterns between the northern North Sea and the English Channel.

Phytoplankton Colour Index, Northern North Sea



Over the most recent decade the PCI for the northern North Sea is 67% higher than in the 1960s, and over the last five years is 29% higher than the mid to late 2000s.

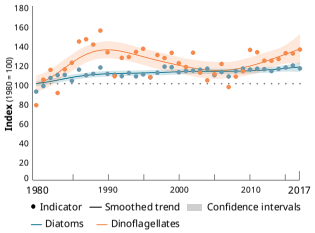
Phytoplankton Colour Index, English Channel



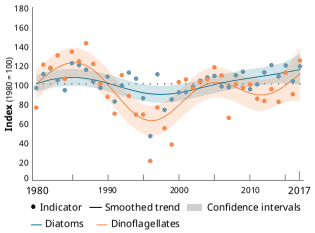
Similarly, in the English Channel the PCI is 95% higher in the last decade and 18% higher in the last five years.

Changes in diatoms and dinoflagellates are associated with changes to trophic pathways and differing roles in the carbon cycle[19]. Increases in the English Channel contrast with periods of decrease and overall stability in the northern North Sea.

Abundance indicator for diatoms and dinoflagellates in the northern North Sea

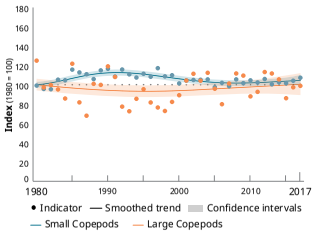


Abundance indicator diatoms and total dinoflagellates in the English Channel

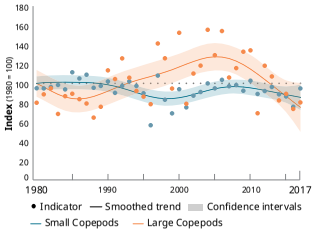


Small copepods are key prey for larval fish and tend to dominate the zooplankton in the southern and central North Sea and so are key trophic links in these areas. Declines in small copepods in the North Sea have been linked to changes in primary productivity[20].

Abundance indicator for small and large copepods in the northern North Sea



Abundance indicator for small and large copepods in the English Channel



Over many decades, the abundance of planktonic larvae – for example, of fish and crustaceans – has increased in most areas associated with increasing temperatures. Although the total abundance of large copepods is more variable, the composition has changed to an increasing dominance of more temperate species as a result of climatic change, highlighting the need to retain taxonomic resolution in monitoring programmes to effectively identify the drivers of change.

RELATED DRIVER OF CHANGE

p54 Marine – Climate change

CLIMATE CHANGE AND OCEAN ACIDIFICATION

Our seas are shaped by complex and interacting climatic factors as a result of human activity and the large oceanographic climatic systems which occur in the marine environment over decadal cycles. Our understanding of specific mechanisms continues to improve and there is clear evidence that climate change is causing rising temperatures which are affecting currents, mixing and salinity as well as increased acidification of the seas. Marine ecosystems are consequently impacted through direct and indirect effects on the distribution and abundance of species groups, including plankton, fish, seabirds and marine mammals, as well as on entire habitats – from the intertidal zone to the deepest parts of the seas.

PRESSURE

Eight of the 10 warmest years for UK sea surface temperature have occurred since 2002.



STATE

Ocean processes have been affected and large-scale shifts in distribution revealed throughout the marine food web.

RESPONSE

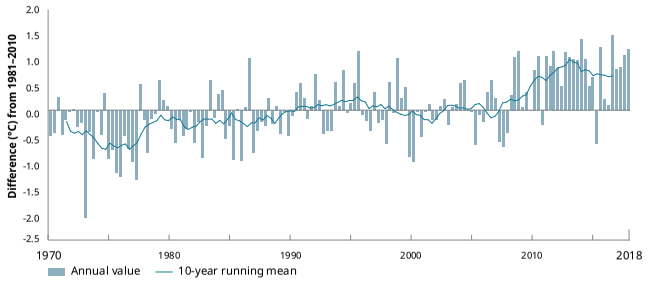
Building resilience in marine systems will help mitigate the effects of climate change on vulnerable species.

Photo: Ernie Janes (rspb-images.com)

The impacts of climate change outlined earlier in this report focused on terrestrial species and habitats. All elements of the marine ecosystem are highly interconnected and climate change causes knock-on effects from primary producers through the food web to predators and even habitats.

ENVIRONMENTAL CHANGES

Time series of average annual sea surface temperatures in °C for UK coastal waters, expressed as anomalies relative to the 1981 to 2010 average[2]



Understanding how the temperature, salinity and pH of our seas are changing is critical, as they all play an important role in marine ecosystem functioning.

- Long-term records show a warming trend in UK waters, despite short-term natural variability. On average, coastal sea surface temperatures have been 0.6°C warmer in the most recent decade compared to the 1961–1990 average[1].

The blue bars show the annual anomalies relative to the 1981–2010 average, shown as the grey horizontal line, and the blue line shows the 10-year running mean.

- The salinity of surface waters to the north and west of the UK has increased since the 1970s. Conversely, salinity in deep waters in the same region decreased between 1960 and 2000, remaining stable in more recent years[3].
- Acidification, caused by the uptake of CO_2, has reduced the pH of waters around Europe, apparently more rapidly so in UK waters than in the North Atlantic as a whole. This has the potential to adversely affect organisms that require calcium carbonate. Acidification is also of particular concern as it could further reduce the rate at which CO_2 is absorbed from the atmosphere, thus aggravating climate change[4].
- On average, the sea level has risen by 16cm since the start of the 20th century, putting increasing pressure on intertidal habitats and increasing the risk of severe flooding and coastal erosion[5].
- Seasonal stratification, which restricts the extent of mixing between shallow, warmer waters and deeper, cooler waters, is occurring earlier on average and lasting longer[6]. Stratification influences plankton growth rates, bloom timing and species composition and distribution, so changes to the timing and strength of mixing have impacts on the distribution and abundance of marine animals that depend on plankton for their food.

Photo: Nick Upton (rspb-images.com)



RESPONSES THROUGH THE FOOD WEB

PHYTOPLANKTON

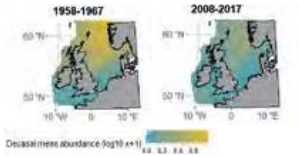
Phytoplankton are at the base of the marine food web. Phytoplankton growth and production can be affected by various factors, including nutrient availability, temperature and light levels. Phytoplankton biomass, as presented in the PCI[7] (page 53), has been used as a proxy for primary production in a number of studies[8,9]. Long-term increases in phytoplankton biomass of 21% and 13% have been reported in the coastal and open North Sea, respectively, between the 1980s and early 2000s[8]. More recently however, estimates of primary production in the North Sea indicate a declining trend as well as changes in species composition and timing of seasonal events, with knock-on effects on zooplankton abundance and fish recruitment[9].

ZOOPLANKTON

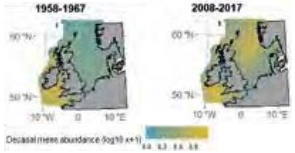
Copepods (small planktonic crustaceans) make up a large proportion of the zooplankton biomass and play a key role in converting energy from primary producers up the food web. Warming temperatures have driven northward shifts in the distribution of zooplankton in the North East Atlantic, from the 1960s to the late 1980s, bringing smaller warm-water copepod species into UK waters[10,11].

An example of this has been the northward shift in the relative proportions of two closely related copepod species, the cold-water *Calanus finmarchicus* and the warmer-water relative *C. helgolandicus* (see map on this page). The implications of these shifts are widespread, not least because of changes to community composition but also because seasonal patterns of production and overall abundance vary between these species. This has consequences for predators of plankton, including fish[12], which in turn can impact on species at higher trophic levels, such as seabirds[13,14,15].

Colder-water species, *Calanus finmarchicus*, moving polewards



Being replaced by warmer-water species, *Calanus helgolandicus*, moving northwards



FISH

Marine fish populations have been influenced by commercial fishing, and in addition, climate change has led to distributional changes[16,17] at rates up to three times faster than those detected for terrestrial species[18]. A large part of the increase in abundance of fish species described in this report has been due to improved fisheries management, allowing some commercially fished species to recover from low baselines. At the same time, 70% of the demersal species studied around the UK have responded to warming sea temperatures by changes in abundance across their distribution[19]. Specifically, this has led to increases in the populations of smaller-bodied, non-commercial species, such as Boarfish in the Celtic Seas[20,21]. For pelagic fish, there is evidence of community change from cold-water assemblages (Herring and Sprat) in the 1960s–80s to warmer-water assemblages (Mackerel, Sardine and Anchovy) from the 1990s onwards, although changes in the distribution of prey, also affected by climate change, may have played a role as well[21,22,17].

Disentangling the drivers of change in the marine environment is complex. The impact of warming seas varies geographically and by species, and by interactions between species. The mismatch of the timing of life history events between predators and prey has the potential to impact fish species through availability of food to larval stages of fish[23]. Impacts of climate change on Cod recruitment via plankton may explain the apparent slow recovery from overfishing and underline that despite recovery in fish stocks, future fisheries management needs to consider climate effects on stock productivity.

TOP PREDATORS

Species at the top of the food web, such as large fish, seabirds and marine mammals, are susceptible to the changes in distribution and timing of seasonal events for plankton and fish species they rely on for food. Sandeels are an important trophic link in the food web, being a key prey species for these predators. There is now good evidence that declines in the abundance and nutritional quality of Sandeels has reduced the breeding success and populations of seabirds such as Kittiwake[24,25,26,27]. The cascade of mechanisms are still being investigated but include the impact of fishing on Sandeel populations as well as increasing sea surface temperature, the timing and strength of ocean stratification, and in mismatch in reproductive timings with availability of prey[28].

Photo: Clement Bardot



Boarfish

Deep-diving seabirds such as Guillemot and Puffin have also been adversely affected, with declines seen at colonies around the North Sea[27], most conspicuously in the Northern Isles[28]. Surface feeding seabirds, or species that depend on them, such as terns, Kittiwakes and Arctic Skuas, are likely to be more sensitive to changes to prey as a result of increasing sea surface temperatures. Indeed, southerly colonies in Europe have already been abandoned or show substantial declines[29].

The impact of climate change on marine mammals remains poorly understood[30]. Changes most likely to be detected will be shifts in distribution of species at the edge of their range as water warms. Species such as the Striped Dolphin may extend their range polewards into UK waters, while the more northerly distributed White-beaked Dolphin may increasingly shift out of UK waters[31].

RESPONSE FOR NATURE

Alongside overarching policies, commitments and action to minimise climate change impacts (by reducing CO_2 emissions to limit temperature rise to no more than 1.5°C) the focus has been to seek to build natural resilience in marine ecosystems through:

- Protecting and recovering sites, habitats and species.
- Creating a well-managed network of habitats to aid the movement of species affected by climate change.
- Reducing the impact of other pressures such as unsustainable fishing[32].

To achieve these things the UK Government has supported the work of the UK Marine Climate Change Impacts Partnership (MCCIP), which includes scientists, government, agencies and NGOs, to provide advice on climate change impacts and a sound evidence base on which to build climate change adaptation plans.

The MCCIP has published a series of report cards[33] on how to manage protected species or habitats known to be vulnerable to climate change impacts. Conservation of these species and habitats will support marine communities, or plants and animals, and store carbon that otherwise would be released into the atmosphere and contribute to climate change effects.

Managing vulnerable habitats

Saltmarsh: over 45,000ha of saltmarsh around the UK coast delivers the natural benefits of carbon sequestration and coastal protection, and supports a wide variety of species. However, saltmarsh is threatened by sea-level rise. Site-based adaptive management plans include interventions such as managed realignment to enhance resilience[34].

Maerl beds: calcareous red seaweeds that form maerl beds, providing homes for diverse communities of plants and animals. They are vulnerable to changes in pH, affecting skeleton formation, and changes in temperature affecting growth and reproduction. Reducing harm from bottom trawling and reducing impacts from freshwater run-off will improve their resilience[35].

Horse mussel beds: large, long-lived and slow-growing bivalve mussels can build up over time to create structurally complex reefs sustaining communities of barnacles, scallops and crabs. As they are very slow to recover after disturbance, reducing or removing human pressures is the most effective method of increasing resilience[36].

Photo: Kevin Sawford (rspb-images.com)



Kittiwake

FISHERIES

Fish populations are complex and precious resources: they are of immense value to the fishing industry and provide food and employment for many people, but they are also an integral part of the marine food web and are vital for the survival of many other species, including seabirds, seals and cetaceans.

As well as directly affecting fish populations, fishing activities have wider impacts on marine species and habitats, including physical damage to the seabed caused by bottom contacting fishing gear and bycatch of seabirds and marine mammals. Bycatch also includes non-target fish species, which in turn introduces the issue of discarding unwanted catch at sea.

In recognition of this, restrictions on fishing practices, effort and equipment use are set, ideally based on scientific evidence, to reduce the environmental damage caused by fishing. Good monitoring information must be available to ensure that decision-making is underpinned by a sound evidence base and takes due account of the trends and status of the marine ecosystem as a whole. Ecosystem-based fisheries management is therefore fundamental to securing the urgent and necessary recovery of marine nature.

PRESSURE

Fishing activity – 33% of quota managed fish stocks are harvested unsustainably.

STATE

Demersal fish communities are recovering from overexploitation in the past, but Good Environmental Status (GES) has not yet been achieved in either the Greater North Sea or the Celtic Seas.

RESPONSE

The UK has a commitment to meet GES for fish and other marine wildlife and habitats by 2020.

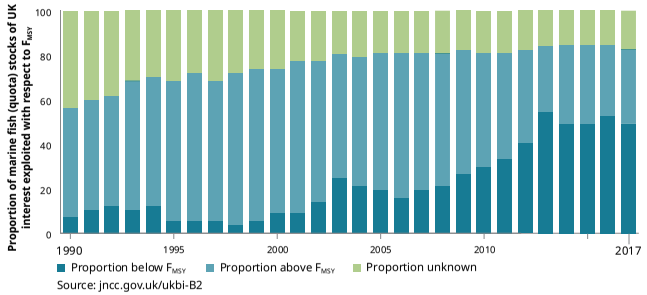
Photo: Chris Gomersall (rspb-images.com)

Marine fish and shellfish are harvested around the UK, representing the most widespread direct human pressure in UK waters[1]. Fish stocks of commercial interest span international boundaries; currently nine nations operate fisheries in the Greater North Sea[2] and 14 in the Celtic Seas[3]. When fish communities are heavily fished, the larger, more profitable fish are removed and the size-mix of the fish stocks is changed. Smaller, less commercially valuable, less reproductive individuals become more dominant, affecting the structure and stability of the ecosystem[4]. Our abundance indicators for marine fish species, both pelagic and demersal, show signs of recovery from a history of overexploitation.

SUSTAINABLE FISHERIES

The UK also has a legal commitment to fish sustainably by 2020 and the assessment of this relies on a measure of the maximum average long-term catch that can be taken from a population without reducing the ability of that population to reproduce itself, termed the Maximum Sustainable Yield (MSY). The official UK Government indicator shows that the percentage of fish stocks fished at or below levels considered to be capable of producing MSY has increased from 7% in 1990 to 49% in 2017, down from a maximum of 54% in 2013, and 33% of quota managed fish stocks are still harvested unsustainably. The UK administration's latest assessment of progress towards Good Environmental Status (GES) under the Marine Strategy Regulations[5] confirmed GES will not be met by 2020 for fish, commercial fish and shellfish, and benthic habitats.

UK Biodiversity Indicator: Proportion of marine fish (quota) stocks of UK interest harvested sustainably



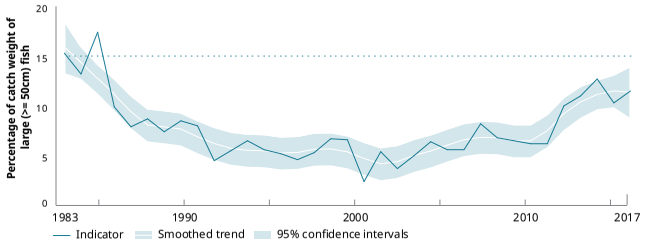
Source: jncc.gov.uk/ukbi-B2

SIZE COMPOSITION OF FISH COMMUNITIES

Indicators of the size composition of fish communities reflect long-term impacts of fishing pressure. One such indicator, the Typical Length Indicator[6], reveals deterioration of the size structure of the fish communities in the North Sea and Celtic Seas between the 1980s and 2000s, such that these communities are now dominated by small-bodied fish. Since 2010 this indicator varies spatially but demersal fish and elasmobranch (shark and ray) communities have shown signs of recovery and the pelagic fish community shows fluctuating trends. The indicator remains low compared to the observed size structure in the early 1980s, and is at record low levels for pelagic and demersal fish in the south-eastern North Sea.

Further evidence for recovery, specifically in the North Sea, is shown by the Large Fish Index. The average percentage of large fish in a catch declined through the 1980s and 90s, to a low of 2% in 2001, but increased subsequently to 12% in 2017. Increases in the indicator could be driven by an increase in a few large commercial fish species[7] and in future is likely to be affected by climate-related changes in species' distributions[8].

UK Biodiversity Indicator: Proportion of large fish (equal to or larger than 50cm), by weight, in the North Sea, 1983 to 2017



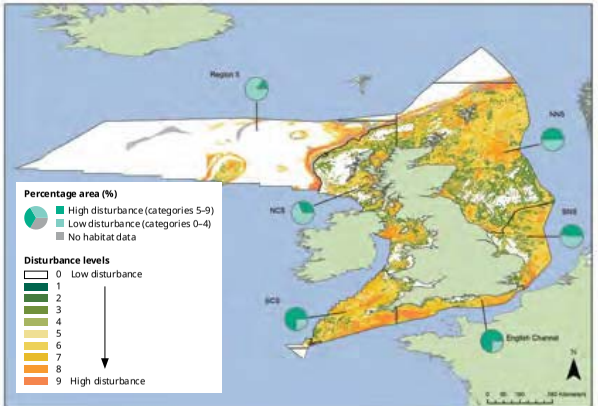
Source: jncc.gov.uk/ukbi-D1a

EXTENT OF PHYSICAL DAMAGE

The most widespread impacts on seafloor habitats occur as a result of frequent use of bottom contacting gear. Physical disturbance can affect seafloor habitats adversely, with shifts in benthic community composition being reported[9]. These shifts are driven by the replacement of larger, long-lived, slow-reproducing species with small, fast-growing species.

In the absence of long-term monitoring datasets to assess the status of marine seafloor species directly, a modelled approach combining habitat sensitivity with impacts from fishing vessels over 12m in length using bottom contacting gear has been used to produce a disturbance map (see opposite)[10]. In this way habitats may be classified as highly disturbed (categories 5–9), either because they are heavily fished or are highly sensitive.

The assessment does not take into account trawling activity prior to 2010 nor the current activity of smaller vessels, so disturbance may be underestimated in some areas. Pressure is modelled evenly across each grid cell so smaller-scale spatial variation in fishing activity may not be represented accurately, which could have an effect on the estimate of overall disturbance.



GSHHG world shore line – Available under GNU Lesser General Public License at www.ngdc.noaa.gov/mgg/shorelines/gshhs.html.

Between 2010 and 2015, the most recent assessment indicates that pressure and disturbance caused by fishing activities are widespread within UK waters, occurring to some degree in 57% of the cells. High-disturbance categories (5–9) accounted for 31% to 75% of the area of assessed UK sub-regional seas, the most disturbed regions being the southern Celtic Seas and the English Channel.

Photo: Laurie Campbell (rspb-images.com)



BYCATCH AND ENTANGLEMENT

Bycatch (the catching of non-target species) can be a serious issue for some species. The most common interactions between cetaceans and fishing gear in the UK involve small cetaceans like Harbour Porpoise and Common Dolphin. It is estimated that 1,500 small cetaceans and 400–600 seals are caught as bycatch each year[11,12,13], as well as an increasing number of cases of Humpback and Minke Whale entanglement off Scotland from ropes to the surface from pots on the seabed[14,15,16].

Annual bycatch mortality for Harbour Porpoise[17], Grey Seal[18] and Common Dolphin[13] in the Celtic Seas and for Humpback Whales around Scotland[15] is assessed to be above precautionary thresholds in relation to population size, suggesting "considerable conservation concern"[19]. Despite this, Grey Seal populations in the Celtic Seas are increasing, thus suggesting wider population mixing between Scottish populations than previously thought[18]. Additional information on bycatch mortality of cetaceans comes from strandings: around 800 cetaceans strand around the UK coast every year, and between 1990 and 2017, 17% of Harbour Porpoise, 43% of Common Dolphin and 36% of Minke Whale strandings were determined as being the result of bycatch[16].

Seabird bycatch hotspots have been identified in areas around the UK where coastal gill nets and other static fishing gear are used, and also in demersal longline fisheries off the west coast of Scotland[11,12]. A risk assessment identified that the seabirds most sensitive to bycatch include species of conservation concern, such as Puffins and Shags[19].

RESPONSE FOR NATURE

REDUCING BYCATCH

The UK has international commitments to reduce cetacean bycatch to as close to zero as possible[20] and yet bycatch of several species remain at levels of considerable conservation concern. Current activities include coordinated stakeholder-led approaches to identify practical, affordable, and effective mitigation and monitoring methods are being developed. These include acoustic and light deterrents, modifications to equipment to improve selectivity and timed closures based on real-time monitoring data[16,21].

Currently between 0.5% and 5% of UK fishing activity is monitored at sea, which varies by activities[16]. Observer programmes are the main source of monitoring data, but other data sources, including cameras, acoustic and satellite options, could be used more effectively to improve monitoring.

Where gill net vessels are equipped with acoustic deterrents known as "pingers", bycatch rates of Harbour Porpoise in UK waters have been much lower, with no clear evidence of habituation, reducing porpoise bycatch by around 15%, to 1,468 individuals in 2014[21].

These problems have been known for over 20 years, and conservation strategies are being developed for cetaceans in each of the UK countries and in 2018 Defra, with the support of the Devolved Administrations, commissioned Joint Nature Conservation Committee (JNCC) to develop a UK Plan of Action (PoA) on Seabird Bycatch, in order to "deliver a coherent approach to understand and where necessary reduce seabird bycatch in UK fisheries". The UK PoA, which is still a work in progress, is in accordance with UK and EU regulations[22,23]. Some best practice already exists, e.g. in the Filey Bay (Yorkshire) inshore gill-net fishery for Sea Trout and Salmon, a combination of a new by-law, regular monitoring, and especially gear modification by introducing higher visibility netting (an innovation of one of the fishermen), has drastically reduced the bycatch of auks but other areas of concern have been identified[24].

DISCARD BAN

A fundamental change in fisheries management has been to tackle the practice of discarding unwanted fish at sea, in some regions accounting for up to 80% of a catch. Not only was this seen as an unacceptable waste of protein with potential economic impacts, but it had significant unintended and negative impacts on some fish stocks as well as distorting marine food webs[25] (which may have included supporting increases in some discard-scavenging seabirds such as Northern Fulmar). The practice was widely condemned and measures under the Common Fisheries Policy introduced a discard ban for commercially harvested stocks which was phased in between 2015 and 2019[26].

However, serious concerns exist that the ban is not being implemented and enforced properly[27,28] and that this could result in overfishing, given levels of unaccounted mortality[29]. Evidence of the continued practice of discarding despite the ban was provided by the DiscardLess project in 2018, stating that "...according to the 2017 data reported to ICES (International Council for the Exploration of the Sea), discards have remained high and landings of undersized fish ... have remained negligible, even for the fish stocks already fully covered by the landing obligation that year"[30].

Best practice avoidance measures to reduce the number of unwanted fish caught, alongside comprehensive remote electronic monitoring with cameras, are vital to avoid overfishing. This will ensure the ban is implemented and will also improve our understanding of the scale of the problem and provide accurate information on the health of fish stocks in order to inform sustainable management and protection of vulnerable species[31].

SUSTAINABLE FISHERIES

While there have been improvements in the sustainability of fisheries over the last decade with North East Atlantic trends for fishing pressure declining and biomass increasing, these positive trends have been challenged in recent years due to a number of factors, including the advent of the discard ban being phased in during 2015–19. Urgent action was advised in the early 2000s to avoid the collapse of the Cod and associated fisheries and harm to the marine environment[32], and while improvements have been made we are once again dealing with the near collapse of Cod stocks in the North Sea[33]. It is clear more needs to be done to ensure all UK fish stocks are diverse, resilient and attain biomass levels that maintain full reproductive capacity. Healthy stocks are vital for a resilient and profitable fishery and for wider marine health.

The UK has a commitment to meet Good Environmental Status (GES) for fish and other marine wildlife and habitats and the pressures impacting upon them by 2020. The recent update of the UK Marine Strategy Part 1 concluded that GES had been achieved for four pressures (e.g. eutrophication and contaminants), was partially achieved for underwater noise and four components of biodiversity and was not achieved for three pressures and three components of biodiversity (including both fish and commercial stocks).

Sustainable fisheries management requires consideration of the broader impacts on habitats, species and trophic interactions, and while these have started to be taken into account through management measures, including through protected areas, progress is slow and often falls short of the ideal. Monitoring and data collection also remains poor, with 12% of stocks being of unknown status[34], including nearly all elasmobranch (shark and ray) stocks; understanding of catches is poor due to low levels of effective monitoring at sea.

PRESSURES ON MARINE NATURE

Fishing and climate change have the most widespread impacts on marine wildlife, but more localised effects come from a range of human activities in the marine environment and these can be categorised into two main categories: 1) pollution, including marine plastics as well as noise and contaminants, and 2) development, including aggregate extraction, port activities and renewable energy developments (e.g. windfarms). Our understanding of the impact of many of these pressures on populations is limited. The spatial distribution and intensity of these activities is important to consider in order to identify hotspots which could be linked to potential environmental pressures.

POLLUTION: MARINE PLASTIC



Photo: Mark Hamblin (rspb-images.com)

The issue of plastics in our seas has recently come to the fore and the United Nations Environment Programme has listed it as a critical problem[1]. Incidents of entanglement and ingestion of plastics by marine animals have been widely reported[2] and our understanding of the scale at which microplastics are found in the oceans is growing[3,4]. One long-running study of accumulated plastics in the stomachs of Fulmars, a widespread seabird that forages exclusively at sea, goes some way to show trends in marine litter. In the North Sea, 93% of beached Northern Fulmars investigated between 2010 and 2014 had ingested some plastic, with an average of 33 particles per bird[5]. This is an increase from fewer than 10 particles per bird in the 1970s and 80s[6,7,8]. The English Channel (68%) and East of England (69%) had the highest plastics load with Scottish Islands the lowest, but still at 58%.

POLLUTION: NOISE

Since 2015, data exist on the levels of impulsive (short, sharp) sound from seismic surveys for oil and gas exploration, explosions and pile driving, as well as some naval sonar. In parts of the northern North Sea seismic surveys may have occurred on 146 days in 2015[9]. Such impulsive noise may cause changes in the behaviour of marine mammals, for example by causing the temporary displacement of Harbour Porpoises[10].

POLLUTION: CONTAMINANTS AND EUTROPHICATION



Photo: Oliver Smart (rspb-images.com)

Eutrophication, which is caused by excessive amounts of nitrogen and phosphorus entering the water, results in algal blooms which can impact on other organisms, deplete oxygen in the sea and reduce overall water quality. Contamination of marine waters with a range of hazardous substances has reduced in recent years and the worst point sources of pollution have been addressed, including a marked reduction in the frequency of oil spills. However, agricultural run-off into marine habitats remains an issue in some local areas and eutrophication was still observed in 7% of assessed areas – mainly in south-eastern parts of the North Sea and in some coastal waters of the Celtic Seas. Discharges of chemicals from the oil and gas industry decreased by 40% between 2009 and 2014[11].

There are concerns about accumulation of persistent organic pollutants in food webs; one particular group of chemicals, polychlorinated biphenyls (PCBs), have been reported at toxicologically significant levels in 40% of UK-stranded porpoises and shown to threaten long-term viability of Killer Whale populations. While other similar chemicals have declined following international bans, PCBs stopped declining in 1998.

DEVELOPMENT: ENERGY



Photo: Ray Kennedy (rspb-images.com)

Oil and gas production declined from a peak in 1999 due to decreasing reserves[12]. In line with international targets to reduce climate change, there has been a substantial increase in renewable energy developments in UK waters and offshore wind now represents two-thirds of new installation energy capacity[13]. There is increasing research into the cumulative impact and long-term effects of windfarm developments, with concerns about collision risk for seabirds and displacement of seabirds and cetaceans from foraging grounds[14,15].

93% OF BEACHED FULMARS IN THE NORTH SEA HAD INGESTED SOME PLASTIC, WITH AN AVERAGE OF 33 PARTICLES PER BIRD.

EUTROPHICATION WAS OBSERVED IN 7% OF ASSESSED AREAS.

RESPONSE FOR NATURE

MARINE PROTECTED AREA NETWORK

The UK has signed up to international agreements to establish an ecologically coherent network of well managed Marine Protected Areas (MPAs) to protect and enhance complex ecosystems and to ensure sustainable use so that the marine environment is safeguarded for future generations to enjoy.

Since 2016, additional sites have been designated, including five Special Areas of Conservation (SACs) for Harbour Porpoise, and management approaches have been adopted in many areas. As of May 2019, in total, the UK network consists of 355 MPAs which encompass 25% of UK waters[16].

However, management measures have been documented in 60% of MPAs but only fully implemented in 10% of sites[17]. In Scotland, 27 MPAs have specific fisheries measures in place and are being developed with the fishing industry and other stakeholders for a further 12 MPAs[18]. Other area-based measures, including voluntary reserves, restricted fisheries areas and safety exclusions zones around offshore windfarms, are considered to provide a contribution to the Scottish MPA network for vulnerable marine ecosystems and specific species, Blue Ling and Sandeels[19]. However, overall these areas would not make a sufficient contribution to making up the shortfall in the MPA network[20].

The effectiveness of MPAs in European waters is less well reported than for tropical systems, so the monitoring of sites closed to human activity, like Lyme Bay, described below, has been seen as a key element in the evidence base to direct UK and European marine conservation management.

CONSERVATION AND SUSTAINABLE FISHING

Lyme Bay has been noted as being an area of "high species richness that includes rare and threatened species"[21], with high-biodiversity reefs formed of mudstone, limestone and chalk. Key species include the Pink Sea Fan, Ross Coral and the commercially fished Great Scallop.

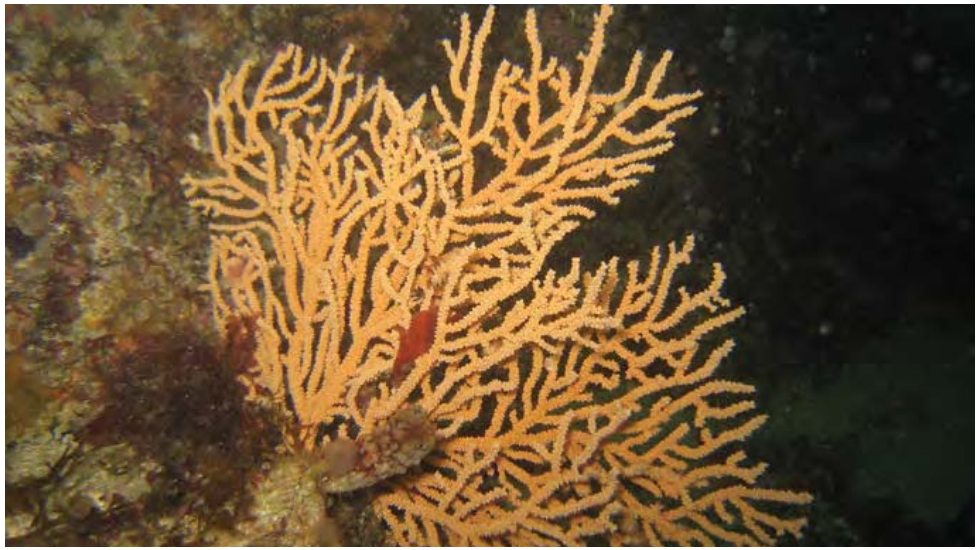
There has been concern for many years about effects of bottom-towed fishing gear on these habitats and the long-term impact on the long-lived, slow-growing reef species[21,22].

In response, voluntary management measures across c22km^2 started in the early 2000s, and Lyme Bay Reefs are now a 236km^2 MPA and SAC. Bottom contact fishing was ended on the reef in 2011 and the area is managed in a collaboration between the local fishing industry, conservationists and scientists to provide benefits for both fishing and conservation.

While the full recovery of marine communities can take many years[23,24], within three years positive responses were seen for species' richness, total abundance and community composition[25]. Recovery was also reported for three indicator species, including the habitat-forming bryozoan, Ross Coral, which plays a key role in formation of the reef and is known to provide the structurally complex habitat essential for juvenile fish[26].

The fishing community benefited from increased income from selling catches under a "Reserve Seafood" label, and increased catches of crabs and scallops, both of which are associated with the protected reef habitat. This is particularly noteworthy for scallops, landings of which have decreased over the same period elsewhere in the UK[27].

Photo: Matthieu Sontag



ENGLAND

England's landscapes have been modified by human activity for millennia. Ever since the clearance of the "wildwood", its associated habitats and eradication of megafauna, biodiversity has undergone major changes. Few, if any, English habitats can be described as truly "wild"; however, human activity has modified and created the semi-natural habitats on which much of the current fauna and flora depend.

Major changes to the landscape have happened through history; for example, the drainage of Fenland started in the 17th century. Through the 20th century the intensification of agriculture has led to loss and fragmentation of semi-natural habitats. Despite this, England still contains a range of internationally important habitats, such as its lowland heathland, ancient woodland and chalk grasslands in the south, the blanket bogs along the Pennines, the coastal estuaries and saltmarshes that provide vital foraging habitat for wintering waterbirds, and the sea cliffs and offshore islands that support internationally important numbers of breeding seabirds.

KEY FINDINGS

1%
decline in average species' abundance.
Our indicator of average species' abundance in England of 241 terrestrial and freshwater species (mainly birds) shows little change since 1970; however, butterflies show significant declines in abundance, while the indicators for birds and mammals show significant increases.

5%
decline in average species' distribution.
Our indicator of average species' distribution in England, covering 5,942 terrestrial and freshwater species over a broad range of taxonomic groups, has fallen by 5% since 1970, and is 1% lower than in 2005.

35%
of species have decreased in abundance.
More species have shown strong or moderate decreases in abundance (35%) than increases (31%) since 1970; likewise more species have decreased in distribution (31%) than increased (24%) since 1970.

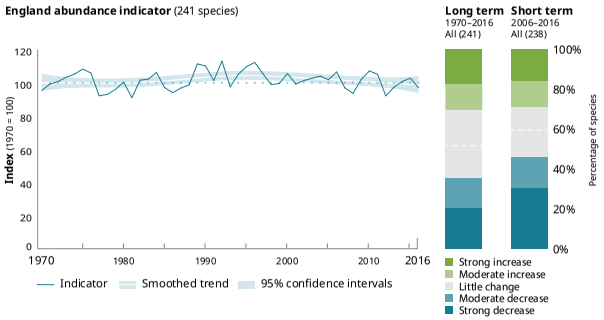
46%
of species show strong changes.
England's wildlife is undergoing rapid change; the proportion of species defined as showing strong changes in abundance, either increasing or decreasing, rose from 38% over the long term to 46% over the short term.

13%
of species are threatened.
Of 7,615 species in England that have been assessed using IUCN Regional Red List criteria, 13% have been classified as threatened with extinction from Great Britain.

Photo: Andrew Mason (rspb-images.com)

COMBINED ABUNDANCE INDICATOR BASED ON ENGLAND-SPECIFIC TRENDS FOR BIRDS (171 SPECIES), BUTTERFLIES (55 SPECIES) AND MAMMALS (15 SPECIES)

It is not appropriate to compare indicator trends between countries as data from different taxonomic groups have been used.



The abundance indicator for 241 terrestrial and freshwater species, for which England-specific data are available, shows a statistically non-significant decline in average abundance of 1% (95% CI -7% to +4%) between 1970 and 2016. Over this long-term period the smoothed indicator fell by 0.03% per year. Over our short-term period, the decline was a statistically non-significant 3%, a rate of 0.29% per year. There was, however, no significant difference in the rate of change between the long and the short term.

The white line with shading shows the smoothed trend and associated 95% CI, the blue line shows the underlying unsmoothed indicator. The bar chart shows the percentage of species within the indicator that have increased, decreased (moderately or strongly) or shown little change in abundance.

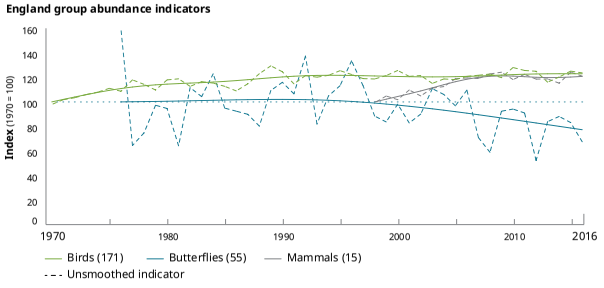
Within multispecies indicators like these there is substantial variation between individual species' trends. To examine this, we have allocated species into trend categories based on the magnitude of population change, over the long and the short-term periods.

- Over the long term, 35% of species showed strong or moderate declines and 31% showed strong or moderate increases; 34% showed little change.
- Over the short term, 46% of species showed strong or moderate declines and 29% showed strong or moderate increases; 25% showed little change.
- Over the long term, 38% of species showed a strong change in abundance (either increase or decrease). Over the short term this rose to 46% of species.

Using a different, binary categorisation of species with positive and negative trends:

- Over the long term, 51% of species showed negative trends and 49% showed positive trends; over the short term, 59% of species showed negative trends and 41% showed positive trends.

ENGLAND-SPECIFIC TRENDS IN ABUNDANCE FOR BIRDS, BUTTERFLIES AND MAMMALS



Based on smoothed trends created using England-specific data:

- The abundance indicator for 171 bird species starts in 1970 and overall shows a statistically significant increase in average abundance of 23% (CI +18% to +28%). Over the short term, the indicator was 2% higher in 2016 compared to 2006. The increase in this indicator is driven by the recovery of species from very low numbers, conservation successes and colonising species, as well as increasing numbers of wintering waterbirds.
- The abundance indicator for 55 butterfly species starts in 1976 and overall shows a statistically significant decline in average abundance of 23% (CI -39% to -6%). Over the short term, the indicator was 15% lower in 2016 compared to 2006.
- The abundance indicator for 15 mammal species starts in 1998 and overall shows a statistically significant increase in average abundance of 21% (CI +17% to +25%). Over the short term, the indicator was 2% higher in 2016 compared to 2006.

CHANGE IN SPECIES' DISTRIBUTION IN ENGLAND

The occupancy indicator for 5,942 terrestrial and freshwater species, for which England-specific trends are available, shows a decline in average distribution of 5% between 1970 and 2015. In 2015 the indicator was 1% lower than in 2005. Because species tend to decline in abundance before they disappear from a site, this change of 5% could reflect more severe underlying abundance declines that we are currently unable to quantify.

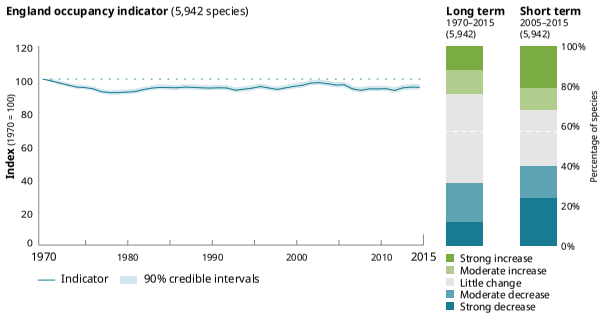
The bar chart shows the percentage of species within the indicator that have increased, decreased (moderately or strongly) or shown little change in distribution (measured as the proportion of occupied sites).

To examine the variation in species' distribution trends, we allocated trends into categories based on the magnitude of distribution change.

- Over the long term, 31% of species showed strong or moderate decreases and 24% showed strong or moderate increases; 45% showed little change.
- Over the short term, 39% of species showed strong or moderate decreases and 32% showed strong or moderate increases; 28% showed little change.
- Over the long term, 23% of species showed a strong change in distribution (either increase or decrease). Over the short term this rose to 45% of species.

Using a different, binary categorisation:

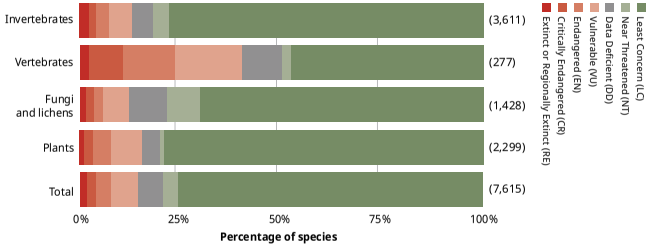
- Over the long term, 57% of species showed negative trends and 43% showed positive trends; over the short term, 57% of species showed negative trends and 43% showed positive trends.



NATIONAL RED LIST ASSESSMENTS FOR ENGLAND

Here we break down the IUCN Red List assessments for Great Britain to show how the proportion of threatened species, based on the number assessed, varies by broad taxonomic group in England. The bars show the percentage of assessed species falling into each of the IUCN Red List categories. Species assessed as Critically Endangered, Endangered or Vulnerable are formally classified as threatened and therefore at risk of extinction.

Percentage of species threatened = (CR + EN + VU)/(total number assessed – DD – RE). The number of species assessed is shown in brackets.



Of the 7,615 species found in England that have been assessed against the IUCN Regional Red List criteria, 971 (13%) of the extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Great Britain (the scale at which Red List assessments are made).

Of the extant terrestrial and freshwater species found in England, assessed using IUCN Regional Red List criteria, 330 plants (15%), 154 fungi and lichens (12%), 105 vertebrates (40%) and 382 invertebrates (11%) are classified as being at risk of extinction from Great Britain.

ENGLAND-SPECIFIC IUCN RED LIST ASSESSMENTS

In order to maximise comparability between taxonomic groups and countries, this report focuses on IUCN Red List assessments undertaken at a Great Britain or whole Ireland level; however, several taxonomic groups have been assessed for extinction risk just within England. These show that:

- Out of 1,859 vascular plant taxa assessed, 36 have been classed as extinct and a further 370 (21%) of extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from England.
- Out of 48 extant terrestrial mammals assessed, for which sufficient data are available, 12 (27%) are formally classified as threatened and therefore at risk of extinction from England.

PRESSURES AND RESPONSES

Nature continues to be under pressure in England. Intensive management of agricultural land, largely driven by policies and incentives since WW2, has been identified as the most significant factor driving species' population change in the UK[1]. As agricultural land constitutes 69% of England's area[2], these changes have had a major detrimental impact on its biodiversity. By 2017, the England farmland bird index had fallen 55% below its 1970 level, while farmland butterflies have declined by 10% on average since 1990[3].

Just 9% of England is wooded. This represents a 45% increase since 1945, but most of this expansion has been through the planting of non-native conifers[4]. Our woodland wildlife is under pressure. In 2016, the breeding woodland bird indicator for England was 24% lower than in 1970, while the indicator for woodland butterflies has fallen by 58% since 1990[3].

Agri-environment schemes (AES) represent the main policy mechanism for reversing widespread declines of farmland wildlife. Research has shown that a targeted approach, such as the Higher Level Stewardship scheme (active between 2005 and 2015), can benefit priority farmland birds[5] and have knock-on benefits for certain wildlife groups; however, their impacts on other species remain unclear[6,7]. In 2017, the total area of land in all higher-level or targeted AES was 1.4 million ha, which represents an increase of 20% over the last decade[3]. However, in order to offset ongoing declines in the UK Government's Farmland Bird Indicator (FBI), it has been estimated that 26–33% of FBI populations would need to be subject to AES-type management[5].

The creation of the EU's LIFE programme brought benefits to England's wildlife; for example, the major wetland habitat creation, restoration and management undertaken through two LIFE projects saw Bitterns increase from 11 booming males in 1997 to 181 in 2018. This habitat creation also benefited colonising wetland species, including invertebrates such as the Fen Wainscot and Marsh Mallow moths, as well as wildlife more generally. The Seabird Recovery LIFE Project in the Isles of Scilly has recently successfully eradicated rats from St Agnes, and both Manx Shearwater and European Storm-petrel have once again bred successfully on the island.

There are many landscape-scale initiatives active in England, funded by NGOs, government agencies or through other sources, like the National Lottery Heritage Fund. Examples include the Great Fen, a 50-year vision to link two fenland reserves in Cambridgeshire; Moors for the Future, which aims to restore and conserve upland habitats in the Peak District and South Pennines; regional forest projects; and the Wallasea Island Wild Coast Project, the UK's largest-ever coastal wetland restoration project. An ambitious collaborative project Back from the Brink was launched in 2017 and aims to save 20 of England's most threatened species from extinction and help a further 200.

Defra launched its 25-year Environment Plan in 2018[8]. This sets out the government's ambitions to help the natural world regain and retain good health in England, including the creation or restoration of 500,000ha of wildlife-rich habitat outside the protected site network.

Photo: Michael Harvey (rspb-images.com)



NORTHERN IRELAND

Northern Ireland's landscape is dominated by agricultural land, which makes up around 75% of the total area[1]. This farmed environment is criss-crossed with a range of special habitats resulting from the wet and mild climate. There are internationally significant areas of blanket bog and large inland and coastal water bodies, including Lough Neagh, the largest freshwater lake in the British Isles, which supports around 100,000 wintering waterbirds, and myriad lakes, fens and raised bogs. Northern Ireland holds species found nowhere else in the UK, including the Irish Hare, Irish Damselfly, Irish Whitebeam, Cryptic Wood White and Pollan. With 650km of coastline, the sea loughs, estuaries and marine environment are a significant component of Northern Ireland's biodiversity.

KEY FINDINGS

66%
increase in average species' abundance of breeding birds.
Our indicator of average species' abundance in Northern Ireland of 41 breeding bird species has increased by 66% since 1994.

38%
decline in average species' abundance of wintering waterbirds.
Our indicator of average species' abundance in Northern Ireland of 36 wintering waterbird species has fallen by 38% since 1988.

43%
decline in average species' abundance of butterflies.
Our indicator of average species' abundance in Northern Ireland of nine butterfly species has fallen by 43% since 2006.

11%
of species are threatened.
Of 2,450 species in Northern Ireland that have been assessed using IUCN Regional Red List criteria, 11% have been classified as threatened with extinction from Ireland as a whole.

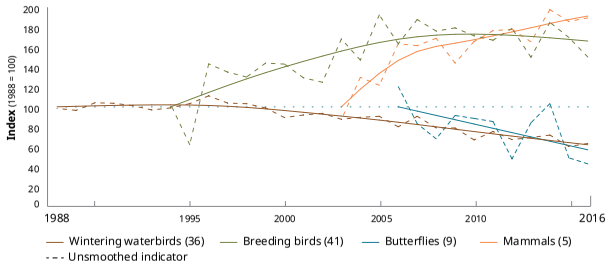
Photo: Andy Hay (rspb-images.com)

NORTHERN IRELAND-SPECIFIC TRENDS IN ABUNDANCE FOR BREEDING BIRDS, WINTERING WATERBIRDS, BUTTERFLIES AND MAMMALS

Due to poor taxonomic coverage of available data, a single combined abundance indicator was not created for Northern Ireland; however, smoothed abundance indicators were calculated for four separate species groups. These were created using Northern Ireland-specific data for all species, except for three of the four bat species which used published trends for all Ireland.

- The abundance indicator for 41 breeding bird species starts in 1994 and overall shows a statistically significant increase in average abundance of 66% (95% CI -45% to +87%). Over the short term, the indicator shows little change in average abundance, being 1% lower.
- The abundance indicator for 36 wintering waterbirds species starts in 1988 and overall shows a statistically significant decline in average abundance of 38% (CI -44% to -33%). Over the short term, the indicator was 27% lower in 2016 compared to 2006.
- The abundance indicator for nine butterfly species starts in 2006 and overall shows a statistically significant decline in average abundance of 43% (CI -67% to -20%).
- The abundance indicator for five mammal species (four bat species and Rabbit) starts in 1998 and overall shows a statistically significant increase in average abundance of 91% (CI +71% to +111%). Over the short term, the indicator was 30% higher in 2016 compared to 2006.

Northern Ireland group abundance indicators



No occupancy-based indicator is currently available for Northern Ireland.

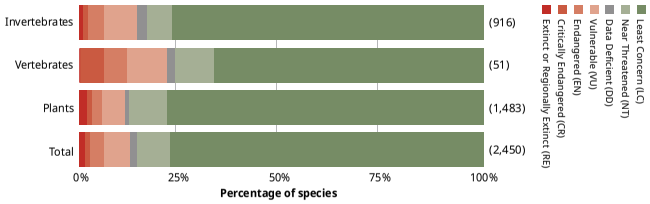


Photo: David Wootton (rspb-images.com)

NATIONAL RED LIST ASSESSMENTS FOR NORTHERN IRELAND

Here we break down the IUCN Red List assessments for the whole of Ireland to show how the proportion of threatened species, based on the number of assessed, varies by broad taxonomic group in Northern Ireland. The bars show the percentage of assessed species falling into each of the IUCN Red List categories. Species assessed as Critically Endangered, Endangered or Vulnerable are formally classified as threatened and therefore at risk of extinction.

Percentage of species threatened = (CR + EN + VU)/(total number assessed – DD – RE). The number of species assessed is shown in brackets.



Of the 2,450 species found in Northern Ireland that have been assessed using the IUCN Regional Red List criteria, 272 (11%) of the extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Ireland as a whole (the scale at which Red List assessments are made).

Of the extant terrestrial and freshwater species found in Northern Ireland, assessed using IUCN Regional Red List criteria, 140 plants (10%), 11 vertebrates (22%) and 121 invertebrates (14%) are classified as being at risk of extinction from Ireland as a whole.

BIRDS OF CONSERVATION CONCERN IN IRELAND

There has also been an all-Ireland bird assessment, which means that Northern Ireland's birds have been assessed alongside those of the Republic of Ireland. The most recent Birds of Conservation Concern in Ireland 2014–2019[2] uses different criteria from the IUCN Red List, assessing each species that breeds or overwinters against a set of objective criteria. These criteria include historical decline, trends in population and range, population size, localisation and international importance, as well as global and European status. Species are placed on the Green, Amber or Red List, indicating an increasing level of conservation concern.

Of the 185 species assessed in an all-Ireland context, 37 (20%) were placed on the Red List, 90 (49%) on the Amber List and 58 (31%) on the Green List. The number of Red-listed species increased by 12 and Amber-listed species by five since the previous review in 2007. New additions to the last Red List include six duck species, as well as a suite of passerines that have undergone population declines and/or range contractions. Populations of breeding waders continue to decline and the long-term future for these species is uncertain.

Photo: Andy Hay (rspb-images.com)



PRESSURES AND RESPONSES

The dominance of farmland within the Northern Ireland landscape means its biodiversity is particularly vulnerable to change in agricultural management. Over recent decades there has been a large-scale move away from mixed farming to a predominantly pastoral system, leading to the loss of semi-natural habitats, overwintering stubbles and hedgerows[1,3]. More than 40% of Northern Ireland's land area now comprises improved grassland[4]. These changes have put pressure on the natural environment and its biodiversity and have led to pollution issues. Northern Ireland produces 12% of the UK's ammonia emissions, mostly from agriculture, while only representing 6% of the land area, and has the greatest percentage of nitrogen-sensitive habitats exceeding critical ammonia levels for both lower and higher plants[5]. Large areas, particularly in County Armagh and County Down, suffer from excessive levels of nitrogen pollution[6] and less than one-third of monitored river water bodies in Northern Ireland were at or above a good standard in 2015[1].

Away from agricultural land, bogs are damaged by peat cutting and heavy grazing, with the latter also affecting heathland[4]. Northern Ireland is one of the least forested regions in Europe and much of what does exist (4.3% of land area) is made up of plantations of non-native Sitka Spruce. These have often been planted in unsuitable areas, such as on blanket bog or in sensitive river catchments, and have had a detrimental impact on important biodiversity. Only 0.04% of the land area comprises ancient woodland[4,7].

Agricultural intensification, driven by UK and European policy, has been identified as the most significant factor driving the decline in species' populations across the UK[8]. Agri-environment schemes (AES) represent the main policy mechanism to turn around losses of farmland wildlife. Since 2018 there has been an increase of nearly 30,000ha in higher-level AES in the country with the launch of the new Environmental Farming Scheme. Prior to this data shows that only 5,000ha were in a higher-level AES in 2017 representing a 90% decline over the last 10 years[9]. Studies have shown mixed results on the impact of AES on biodiversity in Northern Ireland. Assessments of the Environmentally Sensitive Areas scheme identified no benefits for the botanical diversity on moorland or grassland habitats[10], but positive effects for some mammals[11]. Targeted prescriptions, through the Countryside Management Scheme, have been shown to have a positive effect for some priority farmland birds[12].

Photo: Andy Hay (rspb-images.com)



Northern Ireland holds several important seabird colonies. Rathlin Island is the largest; however, biodiversity here is threatened by invasive non-native mammals and the island has been identified as one of the high-priority islands in the UK for eradication of vertebrate INNS[13].

Northern Ireland has no National Parks but Areas of Special Scientific Interest (ASSIs) cover 7% of its total area; this figure incorporates Lough Neagh (395km^2). Data published by the Department of Agriculture, Environment and Rural Affairs (DAERA) show recent declines in both ASSI condition and the proportion of protected terrestrial area under favourable management[14]. Within the marine environment, the total Marine Protected Area increased from 269km^2 in 2009/10 to 2,566km^2 in 2016/17. Management plans for the Northern Ireland MPA network are being developed through the EU funded Marine Protected Area Management and Monitoring (MarPAMM) project to bring these sites into favourable condition[14], as only 4.48% of MPAs in Northern Ireland are currently considered to be under favourable management[15].

Conservation management plans are being produced for the Special Areas of Conservation network funded through the Northern Ireland Rural Development Programme and DAERA's Environment Fund, and by NIEA. Two large EU-funded Interreg projects – Co-operation Across Borders for Biodiversity and Collaborative Action for the Natura Network were launched in 2017. Together these aim to produce 25 conservation action plans for designated sites, restore over 4,500ha of blanket bog and save threatened species including Curlew, Hen Harrier, Irish Damselfly, Marsh Fritillary and White-clawed Crayfish.

SCOTLAND

Scotland holds some of the most diverse landscapes in the UK. From the remote montane habitats of the UK's highest peaks and the extensive expanses of blanket bog and upland heath to the West Atlantic oakwoods, Caledonian pine forests, lochs, coasts and seas, Scotland supports a wide variety of wildlife. The landscapes hold species found nowhere else in the UK, including the Wild Cat, Capercaillie and the endemic Scottish Primrose, Northern February Red Stonefly and Scottish Crossbill[1].

The marine environment is a critical component of Scotland's natural history. The area within 12 nautical miles of the coast is greater than its total land area[1]. The deep seas around Scotland host the UK's only underwater mountains, known as seamounts. Scotland is also recognised as being of international importance for its breeding seabird colonies[2] and marine mammals[1].

KEY FINDINGS

24%
decline in the average species' abundance.
Our indicator of average species' abundance in Scotland of 352 terrestrial and freshwater species has fallen by 24% since 1994. Moths show significant declines in abundance, while the indicators for birds and butterflies have remained broadly stable over time.

14%
decline in average species' distribution.
Our indicator of average species' distribution in Scotland, covering 2,970 terrestrial and freshwater species over a broad range of taxonomic groups, has fallen by 14% since 1970, and is 2% lower than in 2005.

49%
of species have decreased in abundance.
More species have shown strong or moderate decreases in abundance (49%) than increases (28%) since 1994, likewise more species have decreased in distribution (33%) than increased (20%) since 1970.

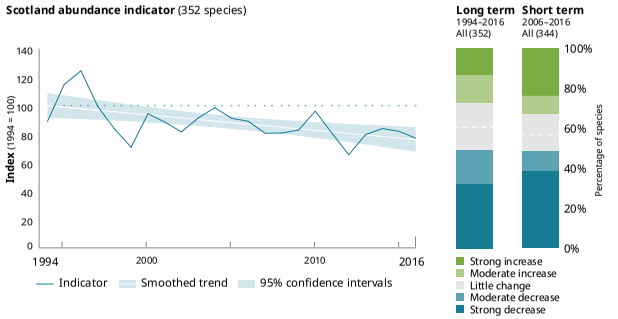
62%
of species show strong changes.
Scotland's wildlife is undergoing rapid change; the proportion of species defined as showing strong changes in abundance, either increasing or decreasing, rose from 45% over the long term to 62% over the short term.

11%
of species are threatened.
Of 6,413 species in Scotland that have been assessed using IUCN Regional Red List criteria, 11% have been classified as threatened with extinction from Great Britain.

Photo: Mark Hamblin (rspb-images.com)

SCOTLAND-SPECIFIC COMBINED ABUNDANCE INDICATOR BASED ON TRENDS OF MOTHS (175 SPECIES), BIRDS (143 SPECIES), BUTTERFLIES (25 SPECIES) AND MAMMALS (9 SPECIES)

Due to poor taxonomic representation before the mid-1990s, the abundance indicator was created from 1994 onwards. It is not appropriate to compare between countries, as data from different taxonomic groups have been used.



The abundance indicator for 352 terrestrial and freshwater species, for which Scotland-specific trends are available, shows a statistically significant decline in average abundance of 24% (95% CI -33% to -15%) between 1994 and 2016. Over this long-term period the smoothed indicator fell by 1.2% per year. Over our short-term period, the decline was a statistically non-significant 12%, a rate of 1.3% per year. There was no significant difference in the rate of change between the long and the short term.

The white line with shading shows the smoothed trend and associated 95% CI, the blue line shows the underlying unsmoothed indicator. The bar chart shows the percentage of species within the indicator that have increased, decreased (moderately or strongly) or shown little change in abundance.

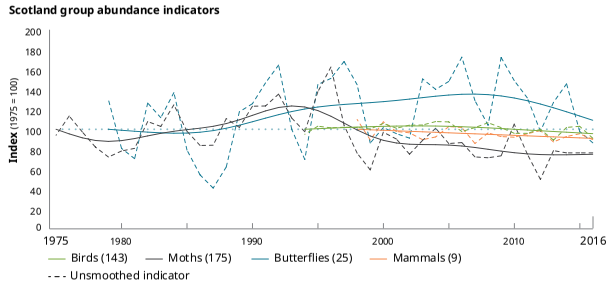
Within multispecies indicators like these there is substantial variation between individual species' trends. To examine this, we have allocated species into trend categories based on the magnitude of population change, over the long- and the short-term periods.

- Over the long term, 49% of species showed strong or moderate declines and 28% showed strong or moderate increases; 24% showed little change.
- Over the short term, 48% of species showed strong or moderate declines and 33% showed strong or moderate increases; 18% showed little change.
- Over the long term, 45% of species showed a strong change in abundance (either increase or decrease). Over the short term this rose to 62% of species.

Using a different, binary categorisation of species with positive and negative trends:

- Over the long term, 60% of species showed negative trends and 40% showed positive trends; over the short term, 56% of species showed negative trends and 44% showed positive trends.

SCOTLAND-SPECIFIC TRENDS IN ABUNDANCE FOR BIRDS, MOTHS, BUTTERFLIES AND MAMMALS



Based on smoothed trends created using Scotland-specific data:

- The abundance indicator for 175 moth species starts in 1975 and overall shows a statistically significant decline in average abundance of 25% (CI -49% to -1%). Over the short term, the indicator was 10% lower in 2016 compared to 2006.
- The abundance indicator for 143 bird species starts in 1994 and has been broadly stable with a statistically non-significant decline in average abundance of 4% (CI -9% to 0%). Over the short term, the indicator was 7% lower in 2016 compared to 2006.
- The abundance indicator for 25 butterfly species has been broadly stable since 1979, with a statistically non-significant increase in average abundance of 9% (CI -27% to +45%). Over the short term, the indicator was 19% lower in 2016 compared to 2006.
- The abundance indicator for nine mammal species starts in 1998 and overall shows a statistically significant decline in average abundance of 9% (CI -14% to -4%). Over the short term, the indicator was 5% lower in 2016 compared to 2006.

CHANGE IN SPECIES' DISTRIBUTION IN SCOTLAND

The occupancy indicator for 2,970 terrestrial and freshwater species, with Scotland-specific data, shows a decline in average distribution of 14% between 1970 and 2015. In 2015 the indicator was 2% lower than in 2005. Because species tend to decline in abundance before they disappear from a site, this change of 14% could reflect larger declines in underlying abundance that we are currently unable to quantify.

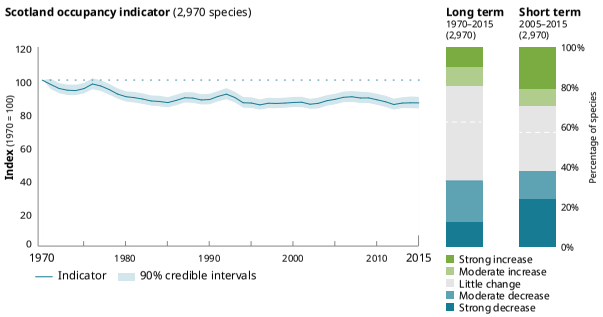
The bar chart shows the percentage of species within the indicator that have increased, decreased (moderately or strongly) or shown no change in distribution (measured as the proportion of occupied sites), based on set thresholds of change.

To examine the variation in species' distribution trends, we allocated trends into categories based on the magnitude of distribution change.

- Over the long term, 33% of species showed strong or moderate decreases and 20% showed strong or moderate increases; 47% showed little change.
- Over the short term, 37% of species showed strong or moderate decreases and 30% showed strong or moderate increases; 33% showed little change.
- Over the long term, 23% of species showed a strong change in distribution (either increase or decrease). Over the short term this rose to 45% of species.

Using a different, binary categorisation:

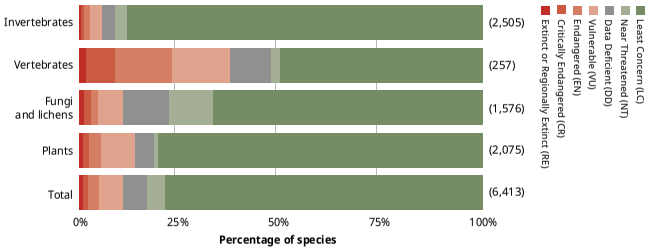
- Over the long term, 62% of species showed negative trends and 38% showed positive trends; over the short term, 57% of species showed negative trends and 43% showed positive trends.



NATIONAL RED LIST ASSESSMENTS FOR SCOTLAND

Here we break down the IUCN Red List assessments for Great Britain to show how the proportion of threatened species, based on the number of assessed, varies by broad taxonomic group in Scotland. The bars show the percentage of assessed species falling into each of the IUCN Red List categories. Species assessed as Critically Endangered, Endangered or Vulnerable are formally classified as threatened and therefore at risk of extinction.

Percentage of species threatened = (CR + EN + VU)/(total number assessed – DD – RE). The number of species assessed is shown in brackets.



Of the 6,413 species found in Scotland that have been assessed using the IUCN Regional Red List criteria, 642 (11%) of the extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Great Britain (the scale at which Red List assessments are made).

Of the extant terrestrial and freshwater species found in Scotland, assessed using modern IUCN Regional Red List criteria, 265 plants (13%), 153 fungi and lichens (11%), 92 vertebrates (37%) and 132 invertebrates (5%) are classified as being at risk of extinction from Great Britain.

PRESSURES AND RESPONSES

Pressure on Scotland's diverse landscapes has resulted in biodiversity losses and gains. There is evidence that some wildlife has fared better in Scotland over recent decades than in the UK as a whole. The woodland bird and farmland bird indicators increased by 69% and 14%, respectively, between 1994 and 2017; while the all-species' butterfly indicator was classed as stable between 1979 and 2017. However, the number of breeding seabirds decreased by 38% between 1986 and 2016[2], with surface-feeders particularly affected, and upland birds declined by 17% between 1994 and 2017[3].

On land, pressures come from many sources, including agriculture, upland management, land-use change, habitat fragmentation, changes in grazing levels, pollution and invasive non-native species (INNS). Climate change places additional burdens on Scotland's wildlife and sometimes exacerbates the impacts from other pressures. Northern species that directly or indirectly depend on cooler climates for survival, e.g. Mountain Ringlet, Cross Whorl Snail and Dotterel, are particularly vulnerable. Some species, including several butterflies, appear to be expanding north due to a warming climate[4].

Scotland's seas are also subject to a range of pressures. Progress has been made on improving water quality, contaminants and eutrophication in coastal waters and some fish stocks are showing signs of recovery. Other pressures, such as those associated with climate change, ocean acidification, marine plastics, fisheries, offshore renewables and other developments, are still challenging and there is evidence of change in pelagic habitats and plankton communities.

INNS continue to impact habitats and native species across Scotland and new incursions continue to occur on its islands. Stoats, not native to the Northern Isles, were found on Orkney in 2010 and rats have reinvaded Handa Island[5,6].

Agricultural land constitutes 68% of Scotland's area[7]. Agri-environment schemes (AES) represent the main policy mechanism to reverse declining farmland wildlife. In 2017, 1 million ha of Scotland were in a higher-level AES[8]. Studies have shown AES do benefit some wildlife groups, particularly when targeted to specific species such as Corncrake. These targeted management prescriptions also benefit wider biodiversity[1,9]. However, general monitoring of biodiversity responses to AES in Scotland has been poorly resourced relative to other UK countries.

The area of woodland in Scotland has more than doubled since the 1940s, and now covers 19% of the land area. Much of this increase is due to the planting of non-native conifers[1,10]. Just 6% of the highly fragmented Caledonian forest remains today and measures are in place to increase this. Cairngorms Connect is an ambitious new partnership with a 200-year vision to enhance habitats, species and ecological processes across the Cairngorms National Park. This focus on native tree species occurs elsewhere in Scotland, e.g. the Central Scotland Green Network.

Invasive species are being tackled on some islands. The Shiant Isles, home to around 10% of the UK's Puffins, were declared rat-free in 2018 following a successful eradication programme. The Orkney Native Wildlife Project was launched in 2018 to eradicate the non-native Stoats and the EU-funded Biodiversity for LIFE project recently started to protect against further non-native incursions by producing biosecurity plans and establishing rapid response hubs.

Over the last 10 years there has been a significant change in marine management, with the introduction of marine planning and progress towards completing the Scottish Marine Protected Area network[11].

Photo: Richard Revels (rspb-images.com)



WALES

From the mountains of Snowdonia and the Brecon Beacons, through enclosed farmland dominated by livestock production, and down the numerous wooded valleys to the estuaries and sea, Wales holds a diverse range of habitats and wildlife. The Welsh Sessile Oak woodlands, regarded as part of the "temperate rainforests" of Europe, hold rich communities of bryophytes, lichens and fungi, while the mountains host rare invertebrates, including the Snowdon Leaf Beetle, and arctic-alpine plants such as Snowdon Lily[1].

The Welsh coastline stretches for over 2,000km. The islands off Pembrokeshire, Anglesey and the Llŷn Peninsula hold seabird colonies of global significance, including the world's largest Manx Shearwater breeding colony and the UK's fourth largest gannetry[2]. Cardigan Bay supports one of the larger semi-resident populations of Bottlenose Dolphin found in the UK.

KEY FINDINGS

52%
decline in the average species' abundance of butterflies.
Our indicator of average species' abundance in Wales of 33 butterfly species has fallen by 52% since 1976; however, widespread breeding birds, wintering waterbirds and mammals (just seven species) show significant increases.

10%
decline in average species' distribution.
Our indicator of average species' distribution in Wales, covering 2,977 terrestrial and freshwater species over a broad range of taxonomic groups, has fallen by 10% since 1970, and is 6% lower than in 2005.

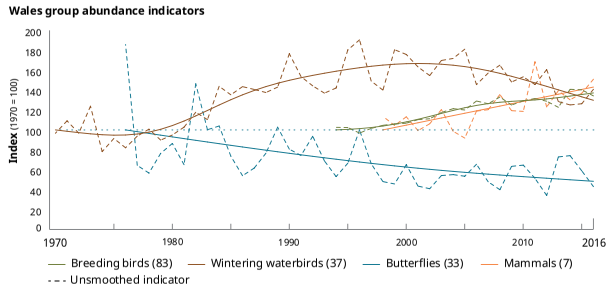
30%
of species have decreased in distribution.
More species have shown strong or moderate decreases in distribution (30%) in Wales than have increased (23%) since 1970.

46%
of species show strong changes.
Wales' wildlife is undergoing rapid change; the proportion of species defined as showing strong changes in distribution, either increasing or decreasing, rose from 24% over the long term to 46% over the short term.

8%
of species are threatened.
Of 6,500 species in Wales that have been assessed using IUCN Regional Red List criteria, 8% have been classified as threatened with extinction from Great Britain.

Photo: Drew Buckley (rspb-images.com)

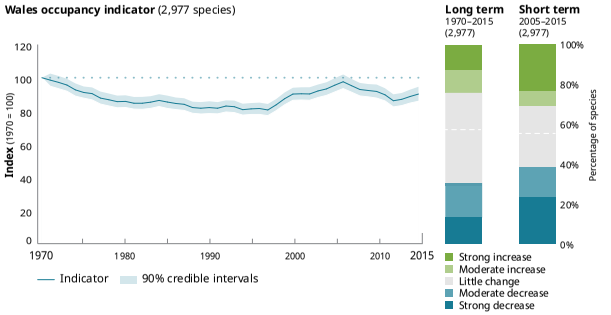
WALES-SPECIFIC ABUNDANCE INDICATORS FOR BREEDING BIRDS, WINTERING WATERBIRDS, BUTTERFLIES AND MAMMALS



Due to poor taxonomic coverage of available data, a single combined abundance indicator was not created for Wales; however, smoothed abundance indicators were calculated, using Wales-specific data, for four separate species groups.

- The abundance indicator for 83 common and widespread breeding bird species starts in 1994 and overall shows a statistically significant increase in average abundance of 37% (95% CI +31% to +43%). Over the short term, the indicator was 10% higher in 2016 compared to 2006. The increase in this indicator is driven by the recovery of a few species from very low numbers, conservation successes and colonising species; importantly, we have insufficient data on some declining species to include them in the indicator.
- The abundance indicator for 37 wintering waterbird species starts in 1970 and overall shows a statistically significant increase in average abundance of 30% (CI +13% to +46%). Over the short term, the indicator was 20% lower in 2016 compared to 2006.
- The abundance indicator for 33 butterfly species starts in 1976 and overall shows a statistically significant decline in average abundance of 52% (CI -69% to -34%). Over the short term, the indicator was 14% lower in 2016 compared to 2006.
- The abundance indicator for seven of 40 mammal species in Wales (six bat species and Rabbit) starts in 1998 and overall shows a statistically significant increase in average abundance of 43% (CI +31% to +55%). Over the short term, the indicator was 20% higher in 2016 compared to 2006.

CHANGE IN SPECIES' DISTRIBUTION IN WALES



The occupancy indicator for 2,977 terrestrial and freshwater species, with Wales-specific data, shows a decline in average distribution of 10% between 1970 and 2015. In 2015 the indicator was 6% lower than in 2005. Because species tend to decline in abundance before they disappear from a site, this change of 10% could reflect more severe underlying abundance declines that we are currently unable to quantify.

The bar chart shows the percentage of species within the indicator that have increased, decreased (moderately or strongly) or shown no change in distribution (measured as the proportion of occupied sites), based on set thresholds of change.

To examine the variation in species' distribution trends, we allocated trends into categories based on the magnitude of distribution change.

- Over the long term, 30% of species showed strong or moderate decreases and 23% showed strong or moderate increases; 47% showed little change.
- Over the short term, 39% of species showed strong or moderate decreases and 30% showed strong or moderate increases; 30% showed little change.
- Over the long term, 24% of species showed a strong change in distribution (either increase or decrease). Over the short term this rose to 46% of species.

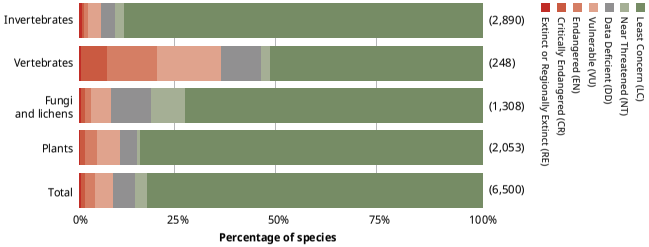
Using a different, binary categorisation of species with positive and negative trends:

- Over the long term, 59% of species showed negative trends and 41% showed positive trends; over the short term, 57% of species showed negative trends and 43% showed positive trends.

NATIONAL RED LIST ASSESSMENTS FOR WALES

Here we break down the IUCN Red List assessments for Great Britain to show how the proportion of threatened species, based on the number of assessed, varies by broad taxonomic group in Wales. The bars show the percentage of assessed species falling into each of the IUCN Red List categories. Species assessed as Critically Endangered, Endangered or Vulnerable are formally classified as threatened and therefore at risk of extinction.

Percentage of species threatened = (CR + EN + VU)/(total number assessed – DD – RE). The number of species assessed is shown in brackets.



Of the 6,500 species found in Wales that have been assessed using the IUCN Regional Red List criteria, 523 (8%) of the extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Great Britain (the scale at which Red List assessments are made).

Of the extant terrestrial and freshwater species found in Wales, assessed using IUCN Regional Red List criteria, 202 plants (10%), 97 fungi and lichens (8%), 86 vertebrates (36%) and 138 invertebrates (5%) are classified as being at risk of extinction from Great Britain.

WALES-SPECIFIC IUCN RED LIST ASSESSMENTS

In order to maximise comparability between taxonomic groups and countries, this report focuses on IUCN Red List assessments undertaken at a Great Britain or whole Ireland level; however, several taxonomic groups have been assessed for extinction risk just within Wales. These show that:

- Out of 1,467 plants assessed, 38 have been classed as extinct and a further 256 (18%) of extant species, for which sufficient data are available, are threatened with extinction from Wales.
- Out of 1,316 lichens assessed, 22 have been classed as extinct and a further 208 (18%) of extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Wales.
- Out of 850 bryophytes assessed, six have been classed as extinct and a further 146 (18%) of extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Wales.
- Out of 225 rust fungi assessed, seven have been classed as extinct and a further 41 (20%) of extant species, for which sufficient data are available, are formally classified as threatened and therefore at risk of extinction from Wales.
- Out of 44 extant terrestrial mammals assessed, for which sufficient data are available, 13 (32%) are formally classified as threatened and therefore at risk of extinction from Wales.

Photo: Graham Eaton (rspb-images.com)

PRESSURES AND RESPONSES

As elsewhere in the UK, nature in Wales is under pressure. Management of agricultural land has been identified as the most significant factor driving species' population change in the UK[3]. With 88% of the Welsh land area utilised for agriculture, nature across the uplands and lowlands has been, and remains, vulnerable to change in farming practices such as grassland and moorland management, livestock type and stocking densities, and a reduction in mixed and arable farming. More than 90% of semi-natural grassland habitats in Wales have been lost since the 1930s[4].

Agri-environment schemes (AES) represent the main policy mechanism to reverse the declines in farmland wildlife. In 2017, 3,500km^2 of Wales was in a higher-level scheme (ESA, Tir Cymen, Tir Gofal or Glastir Advanced), a 21% decline over the previous decade[5].

Evidence to date suggests that Welsh AES have only been partly successful in achieving their biodiversity goals. A recent study points towards Tir Gofal (1999–2012) having positive impacts on wider bird populations[6]. However, other research has pointed towards limited positive benefits for arable plants, grassland fungi, bats, butterflies, Water Voles and Brown Hares[7,8]. Tir Gofal was superseded by Glastir Advanced in 2012.

Again, this has had benefits, such as increases in area and condition of some priority habitat and reduced habitat fragmentation, as well as evidence of increases in generalist woodland and upland breeding birds, but areas of concern remain. These include declines in dwarf shrub heath and the abundance of priority bird species[9].

Around 11% of Wales' land (including much land under agricultural management) is within protected sites for nature, but assessments show that the majority are not in good condition or well managed for the wildlife that depends on them. The Welsh Government's Natural Resources Policy recognises the importance of well-managed protected sites, at the centre of resilient ecological networks, to delivering nature's recovery as well as providing benefits for people.

Woodland cover in Wales has quadrupled to 15% since a low point of 4% in 1918. Most of this increase has been due to the post-World War II planting of non-native conifers. Only 48% of woodlands are considered native and just 14% are classed as ancient and semi-natural. Woodland condition is negatively impacted by grazing pressures from domesticated and wild animals (which can be too little or too much), as well as invasive species, pests and diseases[4].

The EU Celtic Rainforests LIFE project started in 2018 and plans to improve the conservation status of this key woodland habitat, through the control of invasive species and the implementation of active woodland management, grazing and restoration.

Wales' extensive mountains and uplands make an important contribution to its landscape. The Cambrian Mountains have been selected as part of the European Endangered Landscapes Programme. Summit to Sea is a five-year project working with farmers and the local community to bring about ecological, farming and economic benefits in mid-Wales, from the Pumlumon massif down the Dyfi Valley and into Cardigan Bay[10].

The Welsh marine environment is under pressure and the Welsh Government has an obligation to establish a network of Marine Protected Areas (MPAs). To date, 139 MPAs have been designated, covering 69% of Wales' inshore waters[11], but more work is needed to ensure these areas are properly managed for nature. Over recent decades invasive rats have been eradicated from several Welsh islands and a new Biodiversity for LIFE project funded by the EU aims to protect seabirds from future predator incursions by producing biosecurity plans for UK islands, including five in Wales.



Bottlenose Dolphins

THE UK OVERSEAS TERRITORIES AND CROWN DEPENDENCIES

The United Kingdom's responsibilities go far beyond the immediate shores of its constituent countries. The three Crown Dependencies (CDs) lie close to home, while 14 Overseas Territories (OTs) are scattered around the globe. Together they support populations of species of global significance, some found nowhere else on earth.

KEY FINDINGS

Over
32,000
species have been recorded across the OTs, but the actual number of species present is estimated to exceed 100,000 species.

At least
1,549
species are endemic to the OTs. 30% of these are found on St Helena alone.

45
species have become globally Extinct across the OTs and CDs. Most are historic, but three species have formally been assessed as globally Extinct in the 21st century. A further 15 species are classified as Possibly Extinct.

10%
of species are threatened with global extinction.
Of the 5,898 OT and CD species that have been assessed for the global IUCN Red List, 10% are classed as threatened and therefore at risk of global extinction.

40%
of sharks, rays and skates, 36% of reptiles and amphibians, 11% of mammals, 8% of birds and 2% of bony fish found across the OTs and CDs are classified as threatened and therefore at risk of global extinction.

Photo: Alastair Wilson (rspb-images.com)

Wandering Albatross

UNIQUE, GLOBALLY IMPORTANT WILDLIFE

The OTs and CDs hold wildlife populations of global importance. From the wild sub-Antarctic islands of the South Atlantic, to the rainforests of the Caribbean and the tropical islands of the remote Pacific, they hold many unique species and wildlife concentrations found nowhere else in the world.

So far 32,216 native species have been recorded across the OTs; however, information is patchy, and the actual number is estimated to exceed 100,000 species. Many of the OTs are isolated oceanic islands, and as a result they typically hold high numbers of endemic species. At least 1,549 have been documented to date, with 30% found on St Helena alone[1]. This compares to 348 known endemic species in Great Britain[2]. The OTs in the South Atlantic and Antarctic are of global importance for their seabird colonies and contain one third of the world's albatrosses and a quarter of its penguins[1].

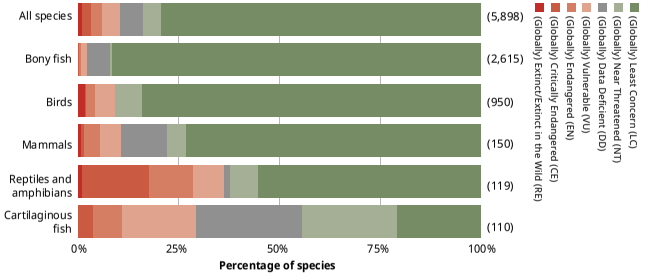
The CDs support a range of wildlife not found in the UK countries. Due to their more southerly location, the Channel Islands have a higher diversity of some wildlife groups, particularly reptiles and amphibians. As islands, the marine environment is an important element of all the CDs.

There are numerous examples of long-term biological studies across the OTs and CDs; however, there is currently insufficient information available to create multispecies indicators, as shown for the UK and some of its component countries.

GLOBALLY THREATENED SPECIES IN THE OTs AND CDs

Here we show the percentage of species found across the OTs and CDs that have been allocated to each of the IUCN Red List categories. All birds and most of the mammals, amphibians, reptiles and bony/cartilaginous fish have now been globally assessed for the IUCN's Red List. More assessments are needed to better understand the true status of other wildlife groups.

Percentage of species threatened = (CR + EN + VU)/(total number assessed – DD – RE). The number of species assessed is shown in brackets.



The IUCN's Red List of Threatened Species represents the world's most comprehensive information source on the global conservation status of species[3]. This documents the global extinction of 45 species across the OTs. Most of these are historic (since 1500 AD) but losses have continued.

Three species were formally assessed as globally Extinct in the 20th century, and three have been added so far this century. A further 15 species are currently classified as Critically Endangered (Possibly Extinct)[3].

Five thousand eight hundred and ninety-eight OT and CD species have now been assessed against the IUCN global Red List criteria. Of these, 560 (10%) of extant species, for which sufficient data are available, are classified as threatened (Critically Endangered, Endangered or Vulnerable), and therefore at risk of global extinction. Of the different taxonomic groups, 40% of cartilaginous fish – sharks, rays and skates – 36% of reptiles and amphibians, 11% of mammals, 8% of birds and 2% of bony fish are assessed as being threatened with global extinction.

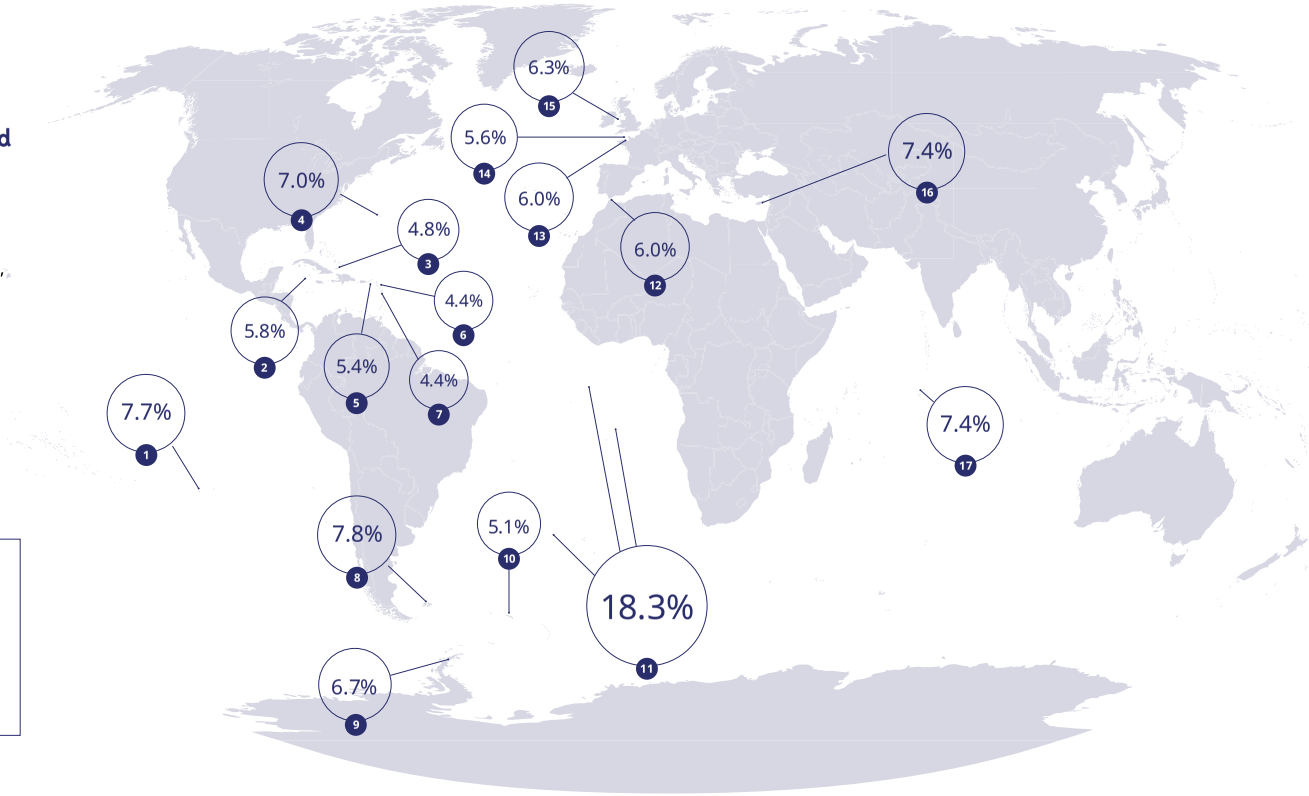
Photo: Alastair Wilson (rspb-images.com)



Bird Island, South Georgia

These pages show the global status of wildlife found across the UK OTs and CDs and a selection of pressures and conservation responses.

Similar proportions of globally threatened species occur across many of the OTs (<10%); however, nearly one in five species on St Helena, Ascension and Tristan da Cunha are formally assessed as being globally threatened with extinction.



The size of the circles denotes the proportion of extant species assessed as globally threatened with extinction on the IUCN Red List (see Methods, page 96).

	OT/CD	Number of species					
		(Globally) Extinct	(Globally) Extinct in the Wild	(Globally) Critically Endangered	(Globally) Endangered	(Globally) Vulnerable	Total number of species assessed for the global IUCN Red List
1	Pitcairn Islands		1	1	10	36	645
2	Cayman Islands	1		20	25	41	1,570
3	Turks and Caicos Islands			7	21	42	1,525
4	Bermuda	5	1	31	16	39	1,313
5	British Virgin Islands			13	27	46	1,681
6	Anguilla	1		7	16	39	1,472
7	Montserrat			10	14	40	1,522
8	Falkland Islands	1		1	15	15	424
9	British Antarctic Territory				2	3	92
10	South Georgia and the South Sandwich Islands				6	7	264
11	St Helena, Ascension and Tristan da Cunha	36	2	58	50	49	955
12	Gibraltar			3	9	33	790
13	Bailiwick of Jersey	1		8	9	30	824
14	Bailiwick of Guernsey	1		8	6	28	789
15	Isle of Man	1		6	7	26	652
16	Cyprus Sovereign Base Areas			3	12	22	547
17	British Indian Ocean Territory			1	11	78	1,258



Photo: Ben Dilley

1 The UK and its OTs are together responsible for the fifth largest area of ocean in the world. The Pitcairn Islands Exclusive Economic Zone covers an area of 834,000km² and was declared a no-take MPA in 2016 as part of the Blue Belt programme (see page 84).

15 The Isle of Man is a hotspot for Basking Shark activity around the British Isles. Here researchers are fitting this globally Vulnerable species with satellite tags to better understand their movements and the pressures they face[4].



Gough Island mouse eating seabird chick

Photo: Ben Dilley

11 Gough Island, within Tristan da Cunha, is one of the most important global seabird colonies; however, introduced mice are having a devastating impact on native wildlife. Recent estimates put the total number of seabird eggs/chicks lost annually to mouse predation at 1,739,000, including the Critically Endangered Tristan Albatross[5]. Gough has been rated as the island with the third greatest need of eradication action globally[6], and an ambitious project is now in development (see goughisland.com).



King Penguin

Photo: Roger Tidman (rspb-images.com)

10 South Georgia, one of the world's last great wildernesses, was officially declared rodent-free in 2018, following the largest global eradication of its type. Introduced rats were having a devastating effect on seabirds and other wildlife.



Black-browed Albatross

Photo: Ben Dilley

10 OTs support some of the world's largest albatross colonies. Incidental bycatch in global fisheries represents the main threat to these birds and ongoing declines in three albatross species on South Georgia have been identified[7]. The Albatross Task Force[8] has dramatically reduced bycatch by up to 99% in some areas and is engaging with the Japanese and Taiwanese high seas fleets, which have the highest overlap with South Georgia albatross, to further reduce these impacts.



Turks and Caicos Rock Iguana

Photo: Sarah Havery

3 The Caribbean OTs hold 17 Critically Endangered reptiles. The Turks and Caicos Rock Iguana is only found on a few small isolated offshore islands and is suffering further losses due to habitat changes and invasive species. A three-year partnership project, funded by the Darwin Initiative, aims to help secure the species' future by establishing effective controls and biosecurity around remaining populations.

11 St Helena's wildlife has developed in extreme isolation and supports at least 502 endemic species. Much of its threatened wildlife is associated with the island's cloud forest[1]. Only small isolated fragments of this habitat remain. A Darwin Plus-funded project is underway and aims to secure the future of this rare habitat and its endemic invertebrates.

13 **14** Although it is extinct on mainland UK, the globally Near Threatened Black-backed Meadow Ant still occurs on Jersey and Guernsey. Here the species is threatened by habitat loss, changes in management and invasive species. Action is being taken to maintain and, where possible, increase the remaining populations. One such project is the reintroduction of grazing on Guernsey's south coast. The species is just about to be legally protected in Jersey.

THE BLUE BELT

The Blue Belt programme is one of the largest conservation initiatives ever undertaken: the UK Government provided almost £20 million for long-term protection for 4 million km² of ocean across the OTs between 2016–2020. Large-scale MPAs have already been designated around St Helena, the British Indian Ocean Territory, the Pitcairn Islands, and South Georgia & the South Sandwich Islands, with further protections being scheduled for Ascension (2019) and Tristan da Cunha (2020).

Over 1.76 million km² of these MPAs have been designated as IUCN Category I fully protected "no-take zones", while other areas permit some sustainable use of marine resources. The fundamental approach of the Blue Belt programme is to work with OT governments and communities to ensure local priorities for marine conservation management are adopted, and that local capacity is built for monitoring and management. Considerable research is underway to set knowledge baselines and to develop appropriate management plans.

Key environmental assets protected within the Blue Belt programme include the largest coral atoll on earth, the Atlantic's second largest Green Turtle nursery and the world's only known aggregation of male and female Whale Sharks, plus the biggest single penguin colony on the planet. A key enabling factor for the development of large-scale MPAs has been satellite surveillance, which can help deter illegal fishing vessels.

HABITAT RESTORATION OPPORTUNITIES FOR THE UK OTs

The extreme 2017 hurricane events in the Caribbean provided clear evidence of the functional, humanitarian and economic value of the natural environment in mitigating inland flooding and coastal zone storm surges. These events highlighted the importance of not only protecting coastal and inland vegetation but also actively intervening to restore key habitats. This is particularly true of coastal mangrove systems that reduce the impacts of hurricane-generated storm surges.

Work by the Joint Nature Conservation Committee (JNCC), in partnership with OT governments and UK-based consultancy Environment Systems, has modelled storm surge episodes in Caribbean OTs and used pre- and post-hurricane satellite data to specifically identify the role of mangroves in mitigating surge impacts on local communities and infrastructure.

In addition to identifying where mangroves have played this mitigating role, "opportunity spaces", where it might be possible to re-establish a particular habitat, have been identified, to help enhance the resilience of the natural capital on the island. This technique has been applied to Anguilla to identify mangrove restoration possibilities and assess the potential role of newly established mangroves in reducing the risk of flooding, to make the case for restoration for economic and disaster resilience purposes.

This trend to intervene to restore and extend important habitats in the OTs is critical to reverse the pattern of degradation and loss.

Photo: RSPB



Pitcairn Island

INVASIVE SPECIES & THE UK OTs

Introduced INNS are one of the primary threats to biodiversity across the OTs and have been implicated in the global extinction of some of their endemic species. Estimates of the impacts of invasive mice on Gough Island World Heritage Site put the number of seabird chicks/eggs predated at 1.7 million per year, but this figure could exceed 3 million. Feral cats threaten the Critically Endangered Turks and Caicos Rock Iguana, while feral pigs have been recorded digging up turtle nests in Montserrat. It is not only invasive mammals that pose a significant threat – introduced insects threaten the only native tree species in Tristan da Cunha, while introduced plants are invading terrestrial habitats in the Falkland Islands.

A variety of approaches are being used to address this threat, starting with the vital work of preventing further introductions by strengthening biosecurity. An EU BEST-funded three-year biosecurity project has been raising public awareness in the Caribbean territories, while a UK Government project has drafted model biosecurity legislation for territory governments to consider and has undertaken a comprehensive programme of horizon-scanning and pathway analysis across all the OTs.

The removal of invasive species is a major undertaking, but many control and eradication schemes are underway. The most significant recent achievement is the clearance of rodents from South Georgia. Here, the spread of other non-native species, such as plants, is also being tackled.

Fundraising and planning for an attempt to restore Gough Island in 2020 continues, to prevent the extinction of the Tristan Albatross and Gough Bunting. At a cost of over £9 million, this is one of the largest conservation interventions planned across the OTs.

↑ Two endemic bird species, the St Helena Plover and Montserrat Oriole, were down-listed from Critically Endangered to globally Vulnerable by the IUCN, due to conservation action; however, threats to both species continue[3].



↓ Four of St Helena's endemic plants, including the St Helena Tea Plant, have been up-listed to Critically Endangered globally on the IUCN Red List since 2015. These species are threatened by habitat fragmentation and very few individual mature plants remain[3].



↓ The endemic St Helena Giant Earwig was declared globally Extinct in 2014, with the last confirmed adult seen in 1967. Habitat degradation and the impact of non-native species is thought to have contributed to its demise[3].



Photo: Roger S Key

↑ The Fin Whale, which has been recorded in the seas around 13 OTs[1], was down-listed from globally Endangered to Vulnerable on the IUCN Red List in 2018. The global population has approximately doubled since the 1970s, but numbers remain well below pre-industrial whaling levels[3].



↓ Five Caribbean OTs' reptiles, including the Sombrero Ameiva, have been up-listed to Critically Endangered globally on the IUCN Red List since 2013. All these species are being affected by introduced non-native species[3].



↓ The St Helena endemic She Cabbage Tree was up-listed to Extinct in the Wild in 2015 following the death of the last wild specimen in 2012. Some cultivated specimens have been replanted in semi-wild situations[3].



↑ THE GOOD NEWS ↓ THE BAD NEWS

CONNECTION TO NATURE

As wildlife populations decline there is increasing concern about people's willingness to act to reverse this. While social, cultural and political factors influence attitudes and behaviour, and there is a recognised gap between people's values and actions, one reason for this lack of engagement is considered to be disconnection from nature: "Simply put, humans don't protect what they don't know and value."[1]

Insights from disciplines such as psychology and sociology indicate the importance of people's "connection to nature" (henceforth "connection") for motivating behaviour to help. The *State of Nature 2016* report highlighted that, while there was regional variation, most UK children had lower connection than desired for motivating conservation behaviour[2,3]. While the future relies on children, teenage and adult connection must be considered for addressing conservation issues now.



Photo: Tony Hamblin (rspb-images.com)

CONNECTION TO NATURE DESCRIPTION

"Connection to nature" describes an individual's relationship with nature and their perception of belonging to the wider natural community[4]. Connection is a complex and multidimensional characteristic. Connection is not developed just through contact and simply getting people outside does not mean they will grow a wildflower meadow or petition for nature[5]. Instead, connection is made up of emotional, cognitive and behavioural aspects – feelings about nature, knowledge and actions.

Sometimes referred to as nature connectedness or relatedness, it is often discussed in terms of disconnection and reconnection[4].

Research has focused on the how, where and why of connection: types of experience that develop connection, relationship with place or psychological response creating the connection[6]. Most research is on the latter[6], with connection expressed as involving feelings of freedom, safety[7], sense of identity[8,9], enjoyment, oneness, empathy and responsibility[7,9-11].

WHY IS PEOPLE'S CONNECTION IMPORTANT FOR NATURE?

One approach for changing someone's behaviour on a subject is to give that subject personal, emotional value. Increasing people's awareness and concern can help instigate actions supporting nature, as recognised by international conservation targets (e.g. Aichi 1). Low connection leaves people with little attachment to what is, or was, present. Consequently, people may not notice biodiversity loss, or, in what's known as "shifting baseline syndrome", only recognise loss within their experience rather than what has gone before[12]. By connecting people, the hope is they will be engaged by current nature, concerned by the declines and more motivated to act, directly or indirectly, for example through supporting stronger government action.



Photo: Tom Simone (rspb-images.com)

While values do not always match behaviour, in children, higher levels of connection are related to greater likelihood of helping nature[3]. In adults, higher connection is also related to higher likelihood of conservation behaviours, such as reducing electricity use[13], and more positive attitudes towards interventions, for example integral swift bricks in homes[14].

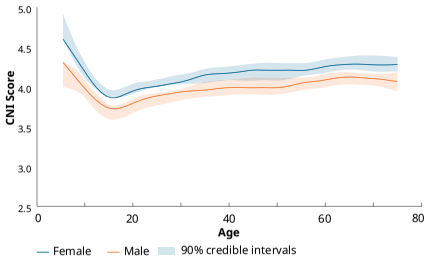
WHO TO CONNECT TO NATURE – RESEARCH SO FAR

Previously, connection was thought fixed in childhood, with studies examining how childhood experiences relate to adult behaviours. For example, how outdoor learning is practiced, its purpose and setting can influence how an ethic of care for nature or understanding of natural processes is developed[15].

However, research from Australia[16] showed that current experiences also increase adult connection. Given this potential flexibility in adulthood connection we need to understand more about connection variation to develop successful interventions. In the UK population, connection differs by gender and age[17].

Females have higher connection scores than males, and connection dips in teenage years[18] (see figure). Thus, developing different interventions for different life stages may improve lifetime connection and behaviour.

Predicted Connection to Nature Index (CNI) scores for ages 5–75 years from generalized additive models (GAMs). CNI scores ranged from 1 to 5



HOW DO WE DEVELOP CONNECTION?

Evidence is building on promising mechanisms for developing connection. One way of initially connecting people is by engaging them with nature in their lives. People often value green spaces because of cultural benefits[19]. Health and well-being benefits can motivate people to take part in nature activities. For example, relaxation is a strong driver for people feeding birds[20] and pleasant green spaces can increase community cohesion[21].

University of Derby research recommends that activities that create emotion, give meaning, showcase nature's beauty, include contact and develop compassion connect people to nature[22]. RSPB research on adults found that, as well as those creating emotion or meaning, activities that involved different senses, included learning and heightened compassion also increased participant's intention to act for nature[23]. Drawing people's attention to details in nature and repeated engagement, key elements of The Wildlife Trusts' 30 Days Wild campaign, help sustain people's connection and maintain behaviour[4].



FUTURE DIRECTIONS – UNDERSTANDING, MONITORING AND EVALUATION FOR CONSERVATION SUCCESS



Photo: Mark Hamblin (rspb-images.com)

Understanding how to influence human behaviour is a priority for conservation. Detailed investigation into what works to connect different audiences across their lifetimes will help long-term conservation action. Measurement of population level trends, for example in England through Natural England's Monitor of Engagement with the Natural Environment surveys[24], are necessary to evaluate large-scale effects. Overall, improving understanding about the relationship between connection and action, how connection varies and consequences for behaviour, is vital for conservation success.

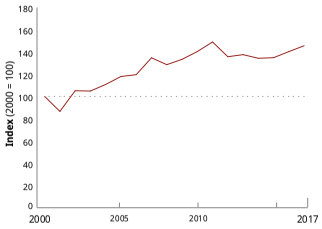
MONITORING THE STATE OF NATURE: WHO, WHAT AND WHY?

This *State of Nature 2019* report, and those published in 2013 and 2016, attempt to assess the state of the UK's nature based on a synthesis of the best available biodiversity data. This would not be possible without the huge effort put into the recording and monitoring of wildlife, most of it done by volunteers.

WHO HELPS MONITOR THE UK'S WILDLIFE?

While professional scientists and conservationists collect much valuable data on biodiversity in the UK themselves, this is outweighed by the huge contribution of volunteers who submit records and take part in structured surveys on a vast range of wildlife. It has been estimated that 18,700 volunteers are involved in structured monitoring schemes that cover bats, birds, butterflies and plants alone, and the financial value of their time contribution has been estimated at £20.5 million per annum[1]. In addition, as many as 70,000 volunteers submit records to national recording schemes and societies (NRSS)[2], or to local environmental records centres (LERCs), for a great range of taxonomic groups. While we do not have precise data on trends in the extent of this volunteer effort, we know that the contributions of time by volunteers (including but not limited to monitoring) to conservation organisations are estimated to have increased by 46% since 2000.

UK Biodiversity Indicator: Index of volunteer time spent in selected UK conservation organisations, 2000 to 2017



Source: jncc.gov.uk/ukbi-A2

The efforts of volunteers reflect the long-standing interest in natural history in the UK, which can be traced back to the expertise of naturalists such as Gilbert White in the late 18th century and John Ray a century before him. Many highly respected experts on the identification and ecology of specific taxonomic groups are volunteers with decades of experience and expertise, without whom recording schemes would not exist.

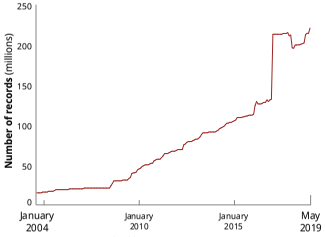
WHAT DO THEY DO?

Broadly speaking, biological data and the schemes that govern its collection can be divided into two categories. Firstly, there are structured surveys that are conducted at predefined sites using a set methodology – such as in the Wider Countryside Butterfly Survey (WCBS)[3]. Data from surveys such as the WCBS are submitted to survey coordinators, increasingly using online forms. Data are analysed using well-established statistical approaches to produce annual trends in abundance, with corrections made to account for biases such as when survey coverage is greater in some regions than others. The robust design and quality assurance procedures of such schemes mean they produce high-quality assessments, and datasets that can be used for numerous research purposes.

The other category is "unstructured" biological records – observations of species at a given time and place that were not collected as part of a structured survey. This means that the methods used and the data collected may vary, and there may be uncontrolled biases associated with the data, for example because observers choose where to go, and so may favour wildlife-rich sites, and are more likely to submit records of rare and special species than more commonplace ones. However, such ad-hoc records cover a huge range of species for which there are insufficient resources, or expert recorders, to run a structured scheme. There are over 90 national recording schemes covering a wide range of taxonomic groups[4] as diverse as slime moulds, stoneworts and leaf-mining moths.

The Biological Records Centre (BRC) curates the datasets compiled by many NRSS, which can be used to map the ranges of species and identify important sites and regions. While structured monitoring schemes remain the "gold-standard", recent statistical developments that account for recording biases[5,6] mean we are now able to use these data to detect trends in the occurrence of species over several decades. These trends play a pivotal role in the *State of Nature* reports, allowing us to report changes in a broad spectrum of the country's wildlife.

UK Biodiversity Indicator: Records added to the NBN, 2004 to 2019



Source: jncc.gov.uk/ukbi-E1

Data from many sources, including the BRC, are made available through the National Biodiversity Network's (NBN) database, the NBN Atlas. The NBN Atlas launched in 2017 and, at the time of writing, holds a remarkable 223,027,119 species' occurrence records, covering 45,448 species in 824 datasets, accessed through the NBN Atlas[7] – and this figure continues to grow rapidly, with jumps caused by the input of new datasets (see above). However, this is still incomplete, with not all data flowing smoothly from surveys, recording schemes, consultancies, scientists and LERCs to the NBN. Furthermore, an unknown but undoubtedly huge volume of data remains in observers' notebooks, photo libraries, social media feeds or simply as fading memories.



Photo: Sue Kennedy (rspb-images.com)

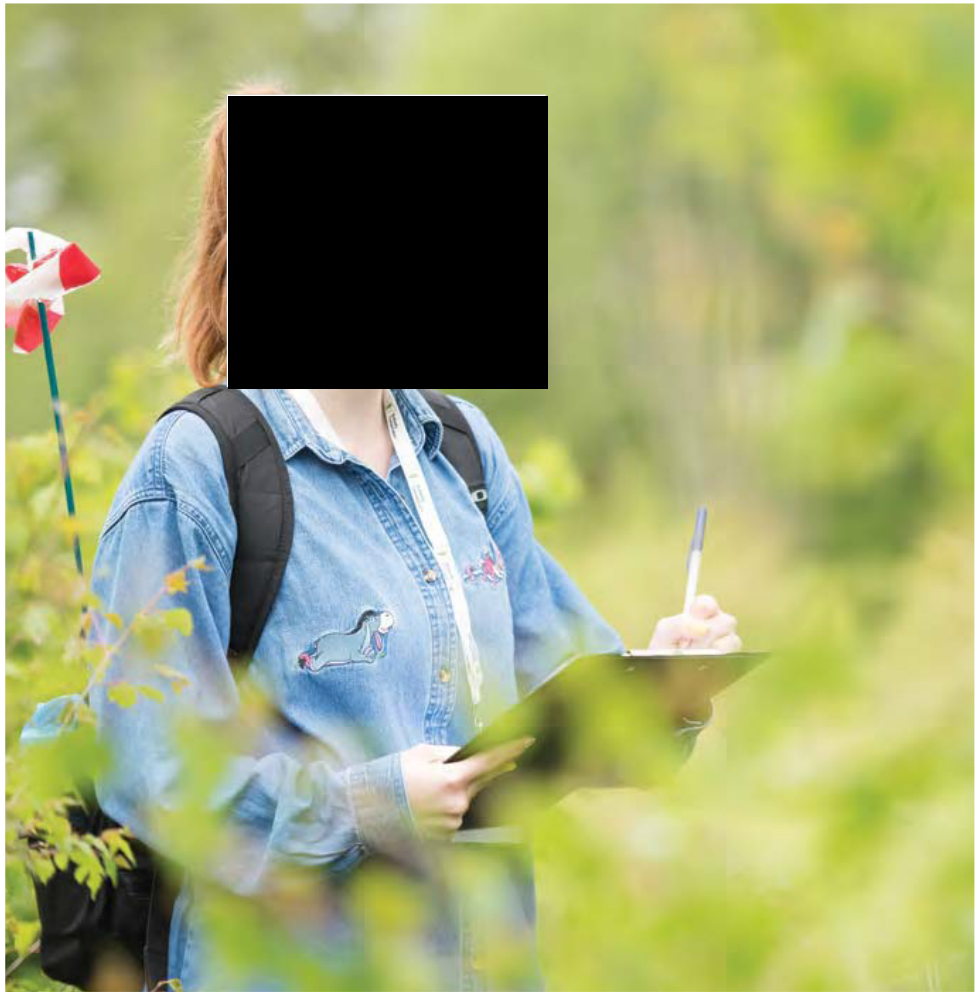
WHY IS THIS MONITORING SO IMPORTANT?

Monitoring builds our knowledge of the natural world, and underpins our efforts to conserve it, and to halt and reverse declines in nature. Measures of abundance and/or distribution, particularly those derived from standardised repeated measurements, allow trends to be calculated and species' status to be determined. Formal assessments such as IUCN Red Lists, which use the best available data to place species in categories of threat using a suite of standardised criteria, are used to identify which species most urgently require conservation attention. Given the squeeze on resources for conservation, such prioritisation is essential for ensuring resources are used efficiently.

Furthermore, repositories of species data are essential tools for spatial planning and defending wildlife from inappropriate development. Datasets on the NBN Atlas and those held by LERCS can aid the identification of the most valuable sites for wildlife, and be used to inform Nature Recovery Network maps. This would enable reserve designation, the targeting of conservation management such as AES and woodland grant schemes, and inform local authorities and developers about sensitive sites and special species.

Finally, by combining data across species, we can look at broader patterns in nature – such as shown by the *State of Nature* headlines, and in the UK Government's biodiversity indicators[8]. Trends in wildlife can tell us about the health of the environment more widely, and what impact human activities are having on it. The reporting of the UK's progress in meeting international targets for biodiversity and sustainability (see page 90) relies heavily on volunteer-collected biodiversity data.

Photo: Ben Andrew (rspb-images.com)



PROGRESS TOWARDS INTERNATIONAL COMMITMENTS: THE AICHI TARGETS

SAVING NATURE GLOBALLY

Recent reports[1] have shown that humankind is failing to halt the loss of biodiversity globally, which continues at an alarming rate. The demands of land-use conversion and other pressures now mean that very few habitats on earth are untouched by people, and 1 million species are at risk of extinction[1]. The recent global report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)[1] issued stark warnings, including that:

- Nature and its vital contributions to people, which together embody biodiversity and ecosystem functions and services, are deteriorating worldwide.
- Direct and indirect drivers of change have accelerated during the past 50 years.
- Goals for conserving and sustainably using nature and achieving sustainability cannot be met by current trajectories, and goals for 2030 and beyond may only be achieved through transformative changes across economic, social, political and technological factors.

The Convention on Biological Diversity (CBD)[2] is a global treaty, opened for signatures in 1992, intended to tackle this crisis. The 10-year Strategic Plan for Biodiversity[3] was adopted in October 2010, and includes five strategic goals underpinned by 20 global biodiversity targets – the Aichi Biodiversity Targets, named after the Japanese prefecture in which they were agreed. Parties to the CBD and other multilateral environmental agreements use these targets to set priorities, develop action plans and establish national targets, and are expected to report on their progress at regular intervals. The most recent of these national reports (the sixth) is being used to create a global assessment of progress, *Global Biodiversity Outlook 5*, which is expected to be published in June 2020. This will be the final report on collective global progress over the last decade, and will signal whether national actions have been sufficient to halt biodiversity loss by 2020. Unfortunately, recent predictions[1] suggest that the majority of the Aichi Biodiversity Targets are unlikely to be met at a global scale.

THE UK'S CONTRIBUTION TO THE GLOBAL PICTURE

The UK's sixth national report to the CBD[4] was submitted in March 2019 and drew upon a huge range of information, such as the UK's Biodiversity Indicators[5] (which feature heavily in the *State of Nature 2019* report), as well as many other robust scientific sources. The report assessed that the UK is on track to meet five of the 20 targets. Although progress has been seen towards another 14, this has not been sufficient to meet the targets; one target was not assessed. In addition to this UK reporting, the Scottish Government has published regular reports on Scottish progress towards the targets[6].

The Aichi Biodiversity Targets are often multifaceted, ambiguous, hard to assess objectively, and sometimes lack clear mechanisms for measuring progress. In the UK, governments, advised by the country nature conservation bodies, are responsible for reporting progress against the targets to the Secretariat of the CBD. There have been disagreements between government and some NGO State of Nature partners regarding both the interpretation of data and of the targets themselves, perhaps inevitably given these conditions. It is notable, however, that these differences centre on a minority of the 20 Aichi targets.

The text on the opposite page focuses on those targets which are central to actions to help biodiversity in the UK, and thus the UK's contribution towards the global effort to help biodiversity. Much of the content of *State of Nature 2019* is relevant to these targets, and tells readers about the UK's progress towards them, adding to the text in the UK's sixth national report to the CBD[4].

LOOKING FORWARD – 2020 AND BEYOND

2020 is a crucial year – a new set of global targets will be negotiated, there is expected to be an uplift in national commitments to tackle climate change and it will be a key milestone for the UN Sustainable Development Goals[7]. These moments and commitments have the potential to deliver increased action and ambition for nature, and to strengthen the connections and solutions between the synergetic issues of nature, climate change and sustainable development.

THE UK GOVERNMENT HAS ASSESSED THAT THE COUNTRY IS ON TRACK TO MEET FIVE OF THE 20 AICHI TARGETS BY 2020.



Extinction of threatened species: "By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained." The *State of Nature 2019* shows that 15% of the 8,418 species assessed are regarded as threatened with extinction in Great Britain, although it is not known how this percentage has changed over time. The abundance of species identified as conservation priorities, including many of the species at greatest risk of extinction, has fallen to 40% of its 1970 value, and continues to fall in the short term (by 22% between 2011 and 2016). The status of species more widely continues to fall in response to a wide range of pressures, with a measure of species' abundance down by 13% over the long term, and one of species' distribution down by 5%. While this report demonstrates how targeted conservation action can prevent extinction, and gives examples of wonderful and inspiring successes, it is clear that more needs to be done to address the needs of threatened species and thus meet this target.



Sustainable management of marine living resource – see pages 58–61. Some UK fish stocks are showing signs of recovery in response to sustainable fisheries measures, but not all stocks are fished at sustainable levels, and issues such as bycatch and trawling damage to seabeds persist.



Sustainable agriculture, aquaculture and forestry – see pages 18–21, 26–27 and 42–45. Incentives, such as AES, have been developed to encourage the sustainability of agriculture, aquaculture and forestry in the UK. However, there has been a recent decline in the area of land under higher-level AES, and critically relevant indicators of wild birds show either no recovery from previous depletion (woodland) or continuing decline (farmland).



Pollution reduced – see pages 38–41. Legislative controls have led to dramatic reductions in pollution from point sources in recent decades. Diffuse pollution in air and water remains a key pressure impacting on the status of species and habitats. Some sources remain above safe levels, however, and their influence on sensitive habitats is still considerable.



Invasive alien species prevented and controlled – see pages 34–37. Systems have been developed, at a Great Britain and all-Ireland scale, to prevent colonisation by INNS, and some progress has been made in tackling established species. However, 10–12 new non-native species establish in Great Britain each year on average, and around one in 10 of these causes adverse impacts.



Protected areas increased and improved – see pages 47 and 63. The UK's protected area network covers 25% of the UK's land area and 24% of its sea area. On land, these totals include landscape designations which do not have the primary purpose of conserving biodiversity and ecosystem services. Terrestrial changes since 1995 mainly reflect the establishment of SACs and SPAs, plus the designation of two National Parks in Scotland in 2002/03. There has been a large increase in the extent of MPAs since 1995, but especially since 2010. The proportion of protected sites assessed as in favourable condition has remained stable (43% of SACs, 50% of SSSIs and 52% of SPAs, in 2019), with an increase in the percentage of site features which are recovering (31%, 35% and 27%, respectively); it is recognised that it will take a long time for species and habitats to recover to favourable condition.



Financial resources from all sources increased – see page 49. There has been a substantial short-term decline in public sector spending on biodiversity in the UK, which has fallen by 29%, from £641 million to £456 million, between 2012/13 and 2017/18. Over the same period, however, the UK's expenditure on biodiversity internationally has increased by 111%, from £97 million to £205 million.

HOW TO INTERPRET THIS REPORT

We have included this section to help you understand the different measures presented in the *State of Nature 2019* report and how they should be interpreted. For full details of the methods and how these measures were calculated, as well as caveats around interpretation, please refer to pages 94–97.

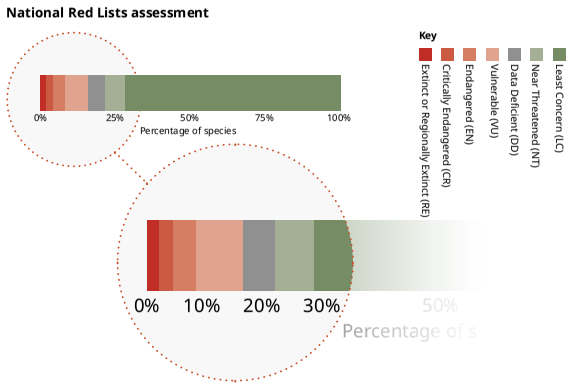
WHAT DATA HAVE WE USED?

- We present trends in abundance (for 697 species) and occupancy (for 6,654 species) for terrestrial and freshwater species across the UK, and trends in abundance (five taxa) for marine species or species groups.
- Abundance trends are based on changes in the number of individuals at a monitored site, a measure that reflects a species' population size. Occupancy trends are based on changes in the number of sites where a species is present. This is a measure that reflects the range of a species. It is usually measured as site occupation at a 1km^2 scale.
- These trends came from a wide range of sources, including national monitoring schemes and biological records.
- Abundance trends are for native species only, but, unless otherwise stated, recent colonist and non-native species were included in the occupancy trends. However, due to the low number of these species, as a result of the time period and record number filtering, their impact on the average trend lines is likely to be minimal[1].
- We present assessments of national Red List status for 8,431 native species.
- Details of our data sources and the species they cover are given in the additional online material.

HOW ARE OCCUPANCY AND ABUNDANCE METRICS RELATED?

The status of species as measured by abundance is considered a key metric for conservation – providing information as to how species are faring and assessing the effectiveness of conservation measures or the impact of particular pressures. However, such data are taxonomically limited and in contrast the volume of opportunistic species' records[2] extends the taxonomic, spatial and temporal coverage of species datasets and analyses. Recent statistical developments have enabled greater use of these datasets for the estimation of species' occupancy trends[3,4,5].

Occupancy and abundance trends are often related, and there is evidence that they tend to operate in the same direction[6,7]. However, the relationship between the two measures of change can be complex. In particular there is evidence that the magnitude of change in occupancy trends is smaller than changes in abundance. This is because many species can show substantial variation in abundance without disappearing from sites or occupying new ones. Additionally, for some species or species groups abundance and occupancy trends move in opposite directions, but this is less common[8,9].



Extinction risk

We summarised the Great Britain Red Lists to present the proportion of species in each threat category overall, and by different taxonomic groups. In each country we interpret existing Great Britain Red Lists, based on those species occurring in a particular country, with the exception of Northern Ireland, where we used all-Ireland Red List assessments. For the OTs and CDs we summarised available global IUCN Red List assessments.

Results reported for each figure include:

- The overall percentage of species assessed that are regarded as threatened with extinction from Great Britain, Ireland or globally. This is the percent of extant species, for which sufficient data are available, classified as Critically Endangered, Endangered or Vulnerable in the latest IUCN Red List assessments.

UK Biodiversity Indicators

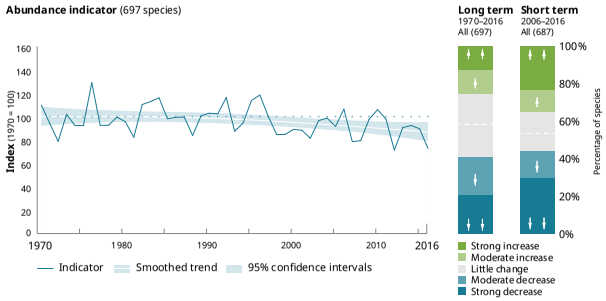
Where appropriate, trend figures from the official UK Biodiversity Indicators[10] are presented to complement the *State of Nature 2019* analyses. The Priority Species Indicator is made up of two parts: one showing change in species' abundance and one showing change in species' occupancy. All other species-based indicators shown are based on trends in abundance.

WHAT ARE THE GRAPHS TELLING ME?

The measures we present, at a UK and individual country level, show the following:

- Change over time – Species Indicator
  - The average change in the status of species, based on abundance or occupancy data.
- Categories of change
  - The percentage of species in each trend category e.g. strong increase or little change.
- Extinction risk
  - An assessment of Red List status for each species occurring in that country.

Please note that our measures are not directly comparable with those presented in the previous *State of Nature* reports because the current report is based on an increased number of species, updated methods and, in some cases, different data sources.



Change over time – Species Indicator

These graphs show indicators based on the abundance data and occupancy data separately.

Species indicator graphs show the average change in the status of species based on either abundance or occupancy data. The shaded areas show the 95% confidence intervals around the indicator line for the abundance trends and 90% credible intervals for the occupancy trends (see page 96).

Results reported for each figure include:

- Total percentage change in the indicator over the long term and the short term.
- Annual percentage change over the long term and the short term.
- To assess if species' status had on average improved or otherwise in recent years, we compared the rate of change in each abundance indicator before the short-term period to the rate of change during the short-term period.

Categories of change

Each species was placed into one of five trend categories based on annual percentage changes.

Results reported for each figure include:

- The percentage of species that showed strong or moderate changes, and those showing little change, in each time period.
- The percentage of species showing strong changes (decreasing and increasing combined) in each time period.
- The overall percentage of species with trends below or above zero in each time period, illustrated by the vertical line across the grey little change segment of the graph.

Thresholds for assigning species' trends to the five categories are given on page 96. A small number of species did not have a short-term assessment as data were unavailable for recent years.

WHAT TIME PERIOD DOES THIS REPORT COVER?

We show abundance trends in species from 1970 to 2016 and occupancy trends from 1970 to 2015. We refer to this as our long-term period. Our short-term period runs from 2006 to 2016 or 2005 to 2015.


Bleeding Tooth Fungus

Photo: Mark Eaton

METHODS

The methods on this page describe the process used to collate measures of species' status and how these were combined into three metrics:

1. **A multispecies indicator, which charts average species' change over time.**
2. **A Categorical Change metric, which describes the proportions of species in five change categories based on the direction and magnitude of average annual change over the long-term and short-term periods.**
3. **A Red List metric, which presents the proportion of species at risk of extinction.**

The methods are based on those used in the second *State of Nature* report and published subsequently in Burns *et al.* (2018)[1].

DATA COLLATION

We collated as many datasets as possible describing population change of native UK species in order to populate the population change metrics (see table on facing page and additional online content). Most of these datasets contained species' time series derived from statistical models, rather than raw counts or observations.

Population change was described either by changes in the relative abundance of species (changes in the number of individuals) or the relative occupancy of species (changes in the number of sites where a species is found, so effectively the change in range of the species). The long-term period was 1970–2016 for abundance time series and 1970–2015 for occupancy time series, due to a time lag in the collation and reporting of biological data. The respective short-term periods were 2006–2016 and 2005–2015.

Data were derived from a wide range of sources; details of the datasets behind our analyses, and the species they cover, are given in the additional online material.

The species' abundance time series included in our assessment met the following criteria:

- Two or more comparable estimates of a species' abundance were made between 1960 and the present, with a broad geographic coverage across the species' UK range.
- Results, or at least the methodology for data collection and/or analysis, had been published.
- Start and end estimates for each species were at least 10 years apart.

If more than one dataset was available for a species, precedence was given to the most robust dataset, based on the survey method subject to the fewest known biases, and maximising the sample size and time period covered. If two or more datasets were of similar quality and duration, then an average was calculated and used.

The occupancy time series were based on the opportunistic recording data collected by NRS; a full list is available in the additional online content. These schemes collect data on a vast array of taxonomic groups, from slime moulds to spiders. However, it can be difficult to use datasets of opportunistic records to assess changes over time, as recording effort varies across the UK and over time.

The majority of the species' occupancy time series included in our assessment were extracted from Outhwaite *et al.* (2019)[2], who used hierarchical occupancy modelling in a Bayesian framework[3,4] to help control biases. They modelled occupancy at a 1km^2 scale, and we retained species time series with a minimum of 10 years of reliable estimates, based on at least 50 records with no more than a 10-year gap in records[5,6].

Raw occurrence records were modelled directly for this report for two taxonomic groups, mammals, using the methods described above, and vascular plants, using an alternative approach, Frescalo (details described on facing page).

In addition to datasets of species' population change, we collated national IUCN Red List assessments (see table).

PROCESSING SPECIES DATA

Within the collated dataset of abundance time series, a small number of time series had missing values, zero values or an end date prior to 2016 and required minor processing following the methods used for previous reports[1]. Data for any years prior to 1970 were removed.

Moth data

Data for 766 moth species were analysed using data from Rothamsted Insect Survey light trap network. The generalised abundance index methodology proposed by Dennis *et al.* (2016)[7] was used to produce UK abundance trends. Four hundred and thirty-two species produced reliable trends based on expert assessment of the underlying data and the analysis results[8].

Summary of the total number of species and the number of species included in each of the UK species' status metrics by taxonomic group. Sources for all contributing datasets are listed in the additional online content.

	Number of species			
	Population change metrics		Red List	UK total
Taxonomic group	Abundance	Occupancy		
FRESHWATER AND TERRESTRIAL				
Birds*	171		241	244
Mammals**	25	20	46	49
Amphibians and reptiles	2			13
Insects***	499 (butterflies and moths)	3,437**	2,773	23,947
Myriapods		50	92	169
Arachnids		402	639	1,560
Crustaceans			51	80
Molluscs		129	192	264
Bryophytes		569	1,055	1,056
Lichens		696	1,662	2,354
Non-lichenised fungi			153	15,195
Vascular Plants		1,351	1,527	1,577
Total	697	6,654	8,431	
MARINE				
Fish	11			
Birds	13			
Mammals	7			
Zooplankton	4 species groups			
Phytoplankton Colour Index	1 species group			
Total	31 species and 5 species groups			

* UK total is regularly occurring breeding and wintering species.
** A proportion of mammal species and moth species were included in both the abundance metrics and the occupancy metrics. See additional online material for species appearing in both metrics.
*** Groups included: aculeates, bugs, beetles, caddisflies, dragonflies, flies, grasshoppers, lacewings, mayflies, moths and stoneflies.

Vascular plant data

For vascular plants, the Botanical Society of Britain and Ireland is the main source of high-quality distribution data, with data holdings stretching back to the 19th century and beyond. Here, we use data from 1930–2018 to derive broad time-period specific estimates of frequency for 1,351 plant taxa. To achieve this, we divided this period into five broad time segments; four of these were retained for analysis, due to the fact that they were considered to be largely unbiased with respect to species' relative frequencies due to Atlas-focused recording activity. We then applied the "Frescalo" algorithm[9] to these data to adjust for variable recording effort within and across time periods. Outputs from this process provided estimates (means and standard deviations) of a species' frequency within each time period. One hundred generalised additive models (GAMs) per species were then fitted to 100 random draws from a species' frequency estimate distribution specified by the Frescalo means and standard deviations. This process enabled us to predict smoothed trends from the 100 GAM fits for a species across all years, while retaining a measure of uncertainty. Finally, the estimated frequency in 1970 was taken as our baseline year, with subsequent values normalised to this point.

To combine these time series with the occupancy time series from other taxa they required conversion to the same scale. The mean and standard deviation from the GAM results were used to generate a distribution of estimates for each species' year combination from which 1,000 random samples were drawn. Estimates were then capped (2, -2) to prevent disproportional contributions from species exhibiting extreme trends and rescaled onto the occupancy scale (0 to 1).

Marine data

Marine fish time series were based on two "bottom trawl" surveys: Greater North Sea International Otter Trawl Q1 and Celtic Seas Scottish Otter Trawl Q1. These datasets have been tidied and refined by Marine Scotland[10,11], giving estimates of abundance per km^2 by length category for each species in each survey visit.

Before creating the population change metrics, for each survey as required we:

- Summarised the average abundance per km^2 for each length category of each species in each year in each survey area.
- Summarised the average abundance per km^2 for each species in each year.
- Retained only species contributing to the first 99% of abundance per km^2 and with a time series spanning at least 10 years, in order to exclude rare or poorly sampled species.
- Coded species as either "demersal" or "pelagic".

The Continuous Plankton Recorder (CPR) Survey dataset provided data for *Calanus* species but also for several zooplankton groups rather than for individual species. Similarly, the PCI gives a representation of ocean colour, which is considered a proxy of the phytoplankton biomass. This means that additional considerations are required in order to interpret how marine biodiversity is changing over time, as a single trend from the CPR may encompass change in a range of individual species.

PRODUCING OUR MEASURES OF SPECIES' POPULATION CHANGE

Separate change metrics were created based on the species' abundance time series and the species' occupancy time series. Previously we have combined these data types into a single metric of change in line with other authors[12]. However, recent research has indicated that the magnitude of these two measures of population change may not be strongly correlated[13,14]. Within the abundance and occupancy datasets the same species contributed to the Species Indicator and the Categorical Change metric.

SPECIES INDICATORS

To create the abundance Species Indicators, all individual species' time series were converted to species indices by expressing each annual estimate as a percent of the first year of the time series and the index was calculated as the geometric mean of the species indices[15]. Species indices starting after 1970 entered the index at the geometric mean value for that year. CI for each Species Indicator were created using bootstrapping across species[16]; in each iteration (N=10,000) a random sample of species was selected with replication and the index was recalculated.

We generated smoothed Species Indicators and associated CI using a generalised additive model[17]. We focus on the total and annual average change in the smoothed Species Indicator in the results and also present the unsmoothed indicator.

Occupancy indicators were created using the methods developed for Indicator C4b of the UK Biodiversity Indicator suite[6] and used the lambda method. This indicator is based on annual growth rates and the credibility intervals are derived from the posterior distribution of the Bayesian occupancy models used to generate the estimates.

The annual average rate of change in each Species Indicator was calculated based on the total change in the long term; the indicator value in its final year, and the short term; the change in the indicator in its last 10 years. If the CI around the final estimate did not contain 100, the long-term change was considered significant. If the CI around the indicator values in the start and end years did not overlap, the short-term change was considered significant.

Testing for change over the period of the indicator

Each abundance Species Indicator was assessed for change between two non-overlapping time periods. Each indicator was modelled using a linear model of the form: ln (index) ~ year + year: Period, where "Period" was a binary variable specifying the non-overlapping short-term and prior (1970–2005) time periods[1].

CATEGORICAL CHANGE

For each species we calculated the total change then the average annual change over the entire long-term period and the recent short-term period – although in many cases the start and/or end years did not match these years exactly. Total change was either a published modelled output or the estimate in a smoothed species index in the penultimate year expressed as a proportion of that in the first year, with exceptions[1]. Each measure of total change was then converted to an annual average rate of change, which was used to categorise species' change. We placed each species into one of five trend categories, defined as follows:

- Strong increase: Annual change greater than or equal to +2.81%, the rate of change that would lead to population size or occupancy doubling or more over 25 years.
- Moderate increase: Annual change between +1.16% and +2.81%.
- Little change: Annual change between -1.14% and +1.16%.
- Moderate decrease: Annual change between -2.73% and -1.14%.
- Strong decrease: Annual change less than or equal to -2.73%, the rate of change that would lead to a population halving or more over 25 years.

This categorisation was based on the magnitude of change, not the statistical significance of that change. Statistical significance is determined by interannual trend variance, which is influenced by sample size, and to the actual interannual variation in population change, which is determined by species' life history. This means that statistical power varies between species and between taxonomic groups. It is common practice to use the magnitude and rate of population change, rather than statistical significance, in order to categorise conservation status assessments. Thus, our values are the best available estimates for each species, but we must acknowledge that many species' trend estimates are highly uncertain.

In addition, we presented a binary split of the proportion of species with positive and negative trends, regardless of magnitude.

IUCN RED LIST ASSESSMENTS

At a global level, the IUCN coordinates the process of assessing which species are threatened with extinction, and has developed Red List assessment criteria[18] to make the process as transparent and consistent as possible.

These criteria are based on a variety of parameters, including the rate of change in species' abundance or occupancy, total population size, number of populations and an assessment of threats.

The IUCN's Red List of Threatened Species represents the world's most comprehensive information source on the global conservation status of species.

Data downloaded from www.iucnredlist.org (21/07/2019) were used to calculate the number of species assessed under the various threat category for each OT and CD. These were based on "Land Region" queries for all OTs and CDs, with the exception of the Bailiwick of Jersey and Guernsey exclusive economic zones, Isle of Man territorial waters, Cyprus Sovereign Base Areas and the British Antarctic Territory, which used spatial queries[19].

How threatened a species is may vary across its range, and often regional or national Red Lists are produced, documenting which species are threatened at different spatial scales[20].

We have brought together all the national Red Lists for Great Britain that have been produced using the latest guidelines from the IUCN[18,20]. For more details of the Red Lists used, please see the additional online material.

We summarised the global and regional Red Lists to present the percent of species in each category and the percent considered threatened across all species, and at different taxonomic levels. We followed recognised guidelines[21] and used the best estimate (the mid-point) when calculating the percent of extant species, for which sufficient data are available, classified as globally Threatened (CR + EN + VU)/(Number assessed – EX – DD).

COUNTRY-LEVEL REPORTING

We do not have the same volume of information on species' trends within the UK's constituent countries as we do for the UK as a whole. We have attempted to repeat analyses, as presented for the UK, in the sections for England, Northern Ireland, Scotland and Wales, but in some cases this was not possible.

We have produced abundance population change metrics (Species Indicators and Categorical Change metrics) across all species for Scotland and England, but only at a taxonomic group level for Wales and Northern Ireland.

Occupancy change metrics were produced using the same methodology as the UK level indicator for England, Scotland and Wales. Insufficient data were available to apply the approach to Northern Ireland or to the species sub-groups used at the UK-wide scale.

Equally, even when metrics have been produced, it should be noted that the population change and distributional change metrics for the constituent countries were based on fewer species and suffered from greater bias towards well-recorded taxa.

For national Red Lists, we used lists of species present in England, Scotland and Wales to interpret the existing Great Britain Red Lists in a national context – this means that the status of a species outside a nation may influence the Red List results presented for that nation. In the case of Northern Ireland, we have used all-Ireland Red List assessments for species occurring in Northern Ireland, as this allowed the consideration of a broader taxonomic scope than data from Northern Ireland alone.

CAVEATS

The datasets presented in this report are a summary of the information available. However, although they cover many species, the datasets have not been selected to reflect a representative sample of UK species, either within or between taxonomic groups or habitats. This means that we should be cautious about extrapolating findings beyond the species assessed.

We have put together datasets collected using different methods, measured at a variety of spatial scales and analysed using different statistical techniques. How a species has been monitored – the method, effort and extent of surveying – can influence whether the results were suitable for our analyses, and indeed the species' trend itself.

Although some rare species are targeted by specific schemes, many of the monitoring schemes that produce the datasets included in this report have a wide range geographically but may not have sufficient sampling density locally to pick up changes in localised or particularly rare species. As a result, trends for relatively few of these species are reported. Our population change metrics may therefore be biased towards the more common, widespread and generalist species, as well as being biased towards certain taxonomic groups. The datasets also differ in spatial coverage. For some species groups estimates are based only on Great Britain and mammal estimates contain some Isle of Man sites. These, however, represent relatively small differences in extent and are likely to only influence the trends of localised species.

We use outputs from hierarchical occupancy models as this method has shown to perform well at dealing with common forms of bias encountered when analysing biological records[4]. As with all modelling approaches, there are several assumptions that should be met when using the approach. Given the number of species included in the analysis, there may be occasions where some of the assumptions are not met, for example intense targeted surveys for certain species may not be fully accounted for in the detection model. However, while the model may not be perfect for all species, it is likely to be better than a model that ignores variation in detectability.

Although official guidelines are used to produce national Red Lists, there is room for variation in interpretation of these guidelines and so there are small differences in the way different authors have compiled the national Red Lists summarised here. This is particularly true in defining which species are not threatened (of Least Concern).

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DRIVERS OF CHANGE

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Action for Conservation
actionforconservation.org

Amphibian and Reptile Conservation (ARC)
arc-trust.org

Association of Local Environmental Records Centres (ALERC)
alerc.org.uk

Badenoch & Strathspey Conservation Group
bscg.org.uk

Bat Conservation Ireland
batconservationireland.org

Bat Conservation Trust (BCT)
bats.org.uk

Biodiversity Ireland
biodiversityireland.ie

Biological Records Centre (BRC)
brc.ac.uk

Botanical Society of Britain and Ireland
bsbi.org

British Arachnological Society (BAS)
britishspiders.org.uk

British Bryological Society (BBS)
britishbryologicalsociety.org.uk

British Dragonfly Society (BDS)
british-dragonflies.org.uk

British Lichen Society
britishlichensociety.org.uk

British Mycological Society (BMS)
britmycolsoc.org.uk

British Pteridological Society (BPS)
ebps.org.uk

British Trust for Ornithology (BTO)
bto.org

Buglife
buglife.org.uk

Bumblebee Conservation Trust
bumblebeeconservation.org

Butterfly Conservation
butterfly-conservation.org

Centre for Ecology & Hydrology (CEH)
ceh.ac.uk

Centre for Environmental Data and Recording (CEDar)
nmni.com/CEDaR

Chartered Institute of Ecology and Environmental Management (CIEEM)
cieem.net

Chester Zoo
chesterzoo.org

Conchological Society of Great Britain and Ireland
conchsoc.org

Continuous Plankton Recorder
cprsurvey.org

Durrell Wildlife Conservation Trust (Durrell)
durrell.org

Earthwatch
earthwatch.org.uk

Freshwater Habitats Trust
freshwaterhabitats.org.uk

Friends of the Earth
friendsoftheearth.uk

Froglife
froglife.org

Isle of Man Government
gov.im

iSpot (The Open University)
iSpotnature.org

Jersey Government Department of the Environment
gov.je/Government/Departments/PlanningEnvironment

John Muir Trust
johnmuirtrust.org

Joint Nature Conservation Committee (JNCC)
jncc.gov.uk

Local Environmental Records Centre Wales
lercwales.org.uk

Mammal Society
mammal.org.uk

Manx BirdLife
manxbirdlife.im

Marine Biological Association (MBA)
mba.ac.uk

Marine Conservation Society
mcsuk.org

Marine Ecosystems Research Programme
marine-ecosystems.org.uk

MARINELife
marine-life.org.uk

National Biodiversity Network (NBN)
nbn.org.uk

National Forum for Biological Recording
nfbr.org.uk

National Trust
nationaltrust.org.uk

National Trust for Scotland
nts.org.uk

Natural England (NE)
gov.uk/government/organisations/natural-england

Natural History Museum
nhm.ac.uk

Natural Resources Wales (NRW)
naturalresources.wales

Northern Ireland Bat Group
bats-ni.org.uk

Northern Ireland Environment Agency
daera-ni.gov.uk/northern-ireland-environment-agency

ORCA
orcaweb.org.uk

People's Trust for Endangered Species (PTES)
ptes.org

Plantlife
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Royal Botanic Gardens, Edinburgh
rbge.org.uk

Royal Botanic Gardens, Kew
kew.org

Royal Society for the Protection of Birds (RSPB)
rspb.org.uk

Royal Zoological Society of Scotland
rzss.org.uk

Scottish Badgers
scottishbadgers.org.uk

Scottish Environment Link
scotlink.org

Scottish Natural Heritage (SNH)
nature.scot

Scottish Wild Land Group
swlg.org.uk

Scottish Wildlife Trust
scottishwildlifetrust.org.uk

Shark Trust
sharktrust.org

States of Guernsey
gov.gg

Trees for Life
treesforlife.org.uk

Ulster Wildlife Trust
ulsterwildlife.org

University of Plymouth
plymouth.ac.uk/research/institutes/marine-institute

University of Sheffield
sheffield.ac.uk

Vincent Wildlife Trust
vwt.org.uk

Whale and Dolphin Conservation (WDC)
uk.whales.org

Wildfowl & Wetlands Trust (WWT)
wwt.org.uk

Wildlife Trusts
wildlifetrusts.org

Woodland Trust
woodlandtrust.org.uk

WWF
wwf.org.uk

Zoological Society of London (ZSL)
zsl.org



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A Focus On Nature
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amphibian and reptile conservation
ALERC
Bat Conservation Ireland
Bat Conservation Trust
National Biodiversity Data Centre
BRC
Botanical Society of Britain & Ireland
British Arachnological Society
BDS
bms
BTO Looking out for birds
buglife
Bumblebee Conservation Trust
Butterfly Conservation
Saving butterflies, moths and our environment
CEH Centre for Ecology & Hydrology
NATURAL ENVIRONMENT RESEARCH COUNCIL
CEDaR Centre for Environmental Data and Recording
CIEEM
CHESTER ZOO
The Conchological Society
DURRELL
EARTHWATCH INSTITUTE
Freshwater Habitats Trust
Friends of the Earth
froglife
Isle of Man Government
iSpot ispotnature.org
Government of JERSEY
JOHN MUIR TRUST
JNCC
THE Mammal SOCIETY
Manx BirdLife
THE MARINE BIOLOGICAL ASSOCIATION
marine
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NBN National Biodiversity Network
NFBR
National Trust
NATIONAL TRUST for SCOTLAND
NATURAL ENGLAND
NATURAL HISTORY MUSEUM
Cyfoeth Naturiol Cymru Natural Resources Wales
Northern Ireland Bat Group
NIEA Northern Ireland Environment Agency
Agriculture, Environment and Rural Affairs
ORCA Looking out for Whales and Dolphins
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Trees for Life
Ulster Wildlife
UNIVERSITY OF PLYMOUTH
The University Of Sheffield.
VWT Vincent Wildlife Trust
WDC WHALE AND DOLPHIN CONSERVATION
WWT
wildlife TRUSTS
WOODLAND TRUST
WWF
ZSL LET'S WORK FOR WILDLIFE

www.nbn.org.uk/stateofnature2019

HM Government

A Green Future: Our 25 Year Plan to Improve the Environment



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Foreword from the Prime Minister



10 DOWNING STREET
LONDON SW1A 2AA

Our natural environment is our most precious inheritance. The United Kingdom is blessed with a wonderful variety of natural landscapes and habitats and our 25 Year Environment Plan sets out our comprehensive and long-term approach to protecting and enhancing them in England for the next generation.

Its goals are simple: cleaner air and water; plants and animals which are thriving; and a cleaner, greener country for us all. We have already taken huge strides to improve environmental protections, from banning microbeads which harm our marine life to improving the quality of the air we breathe to improving standards of animal welfare. This plan sets out the further action we will take.

By using our land more sustainably and creating new habitats for wildlife, including by planting more trees, we can arrest the decline in native species and improve our biodiversity. By tackling the scourge of waste plastic we can make our oceans cleaner and healthier. Connecting more people with the environment will promote greater well-being. And by making the most of emerging technologies, we can build a cleaner, greener country and reap the economic rewards of the clean growth revolution.

The success of the 5p plastic bag charge in reducing the use of carrier bags by 83% shows the difference which government action can make, and demonstrates that protecting our environment is a job for each one of us.

This is a national plan of action, with international ambition. As well as setting an example for others to follow in our treatment of the countryside, rivers, coastlines and air, we will also bring the United Kingdom's international influence to bear in pursuit of a cleaner and safer world.

From reducing our carbon emissions and building resilience against the extreme weather associated with climate change, to leading international action to protect endangered species, the UK is an international champion for the protection of our planet and we will build on our record in the years ahead.

When the United Kingdom leaves the European Union, control of important areas of environmental policy will return to these shores. We will use this opportunity to strengthen and enhance the protections our countryside, rivers, coastline and wildlife habitats enjoy, and develop new methods of agricultural and fisheries support which put the environment first.

We hold our natural environment in trust for the next generation. By implementing the measures in this ambitious plan, ours can become the first generation to leave that environment in a better state than we found it and pass on to the next generation a natural environment protected and enhanced for the future.



The Prime Minister

Foreword from the Secretary of State

It is this Government's ambition to leave our environment in a better state than we found it. We have made significant progress but there is much more to be done. The 25 Year Environment Plan that we have published today outlines the steps we propose to take to achieve our ambition.

Environment is – at its roots – another word for nature, for the planet that sustains us, the life on earth that inspires wonder and reverence, the places dear to us we wish to protect and preserve. We value those landscapes and coastlines as goods in themselves, places of beauty which nurture and support all forms of wildlife.

Respecting nature's intrinsic value, and the value of all life, is critical to our mission. For this reason we safeguard cherished landscapes from economic exploitation, protect the welfare of sentient animals and strive to preserve endangered woodland and plant life, not to mention the greening of our urban environments.

But we also draw from the planet all the raw materials we need to live – food, water, air and energy for growth. So protecting and enhancing the environment, as this Plan lays out, is about more than respecting nature. It is critical if the next generation is to flourish, with abundant natural resources to draw on, that we look after our and their inheritance wisely.

We need to replenish depleted soil, plant trees, support wetlands and peatlands, rid seas and rivers of rubbish, reduce greenhouse gas emissions, cleanse the air of pollutants, develop cleaner, sustainable energy and protect threatened species and habitats.

Previous Governments, here and in other nations, have made welcome strides and driven environmental improvement. Yet as this 25 Year Plan makes clear, there is much more still to do. We must tread more lightly on our planet, using resources more wisely and radically reducing the waste we generate. Waste is choking our oceans and despoiling our landscapes as well as contributing to greenhouse gas emissions and scarring habitats. The success of the 5p plastic bag charge in reducing the use of carrier bags by 83% shows the difference which government action can make, and demonstrates that protecting our environment is a job for each one of us.

The Plan outlines ways to reduce the use of plastics that contribute to pollution, and broader steps to encourage recycling and the more thoughtful use of resources. Over the lifetime of this Plan, we want to eliminate all avoidable plastic waste.

The Government's Clean Growth Strategy – the sister document to this Environment Plan – sets out how we will deliver the clean, green growth needed to combat global warming.

We will do what is necessary to adapt to the effects of a changing climate, improving the resilience of our infrastructure, housing and natural environment.

Population growth and economic development will mean more demand for housing and this Government is committed to building many more homes. However, we will ensure that we support development and the environment by embedding the principle that new development should result in net environmental gain – with neglected or degraded land returned to health and habitats for wildlife restored or created.

Most of our land is used, however, for agriculture not housing. The new system of support that we will bring in for farmers – true friends of the earth, who recognise that a care for land is crucial to future rural prosperity – will have environmental enhancement at its heart.

We will support farmers to turn over fields to meadows rich in herbs and wildflowers, plant more trees, restore habitats for endangered species, recover soil fertility and attract wildlife back. We will ensure broader landscapes are transformed by connecting habitats into larger corridors for wildlife, as recommended by Sir John Lawton in his official review.

Our plan for a new Northern Forest, to which we are contributing more than £5 million, will be accompanied by a new review of National Parks and Areas of Outstanding Natural Beauty.

Planting more trees provides not just new habitats for wildlife – it also helps reduce carbon dioxide levels and can reduce flood risk.

We will work with nature to protect communities from flooding, slowing rivers and creating and sustaining more wetlands to reduce flood risk and offer valuable habitats.

Beyond our coastlines, we must do more to protect the seas around us and marine wildlife. Leaving the EU means taking back control of the waters around these islands. We will develop a fishing policy that ensures seas return to health and fish stocks are replenished. We will also extend the marine protected areas around our coasts so that these stretches of environmentally precious maritime heritage have the best possible protection.

Internationally, we will lead the fight against climate change, invest to prevent wildlife crime, pursue a ban on sales of ivory, and strengthen partnerships to tackle illegal wildlife trade beyond borders, including investigating the feasibility of an anti-poaching taskforce.

We will underpin all this action with a comprehensive set of environmental principles. To ensure strong governance, we will consult on plans to set up a world-leading environmental watchdog, an independent, statutory body, to hold Government to account for upholding environmental standards. We will regularly update this Plan to reflect the changing nature of the environment.

While this 25 Year Environment Plan relates only to areas for which HMG is responsible, we will continue to work with the devolved administrations on our shared goal of protecting our natural heritage.

These actions will, we hope, ensure that this country is recognised as the leading global champion of a greener, healthier, more sustainable future for the next generation.

The Rt Hon Michael Gove MP

[redacted]

Secretary of State for Environment, Food and Rural Affairs



Executive summary

This 25 Year Environment Plan sets out government action to help the natural world regain and retain good health. It aims to deliver cleaner air and water in our cities and rural landscapes, protect threatened species and provide richer wildlife habitats. It calls for an approach to agriculture, forestry, land use and fishing that puts the environment first.

The Plan looks forward to delivering a Green Brexit – seizing this once-in-a-lifetime chance to reform our agriculture and fisheries management, how we restore nature, and how we care for our land, our rivers and our seas.

Our ambitious proposals will tackle the growing problems of waste and soil degradation – issues that affect our urban areas as well as our countryside. They seek to improve social justice by tackling the pollution suffered by those living in less favourable areas, and by opening up the mental and physical health benefits of the natural world to people from the widest possible range of ages and backgrounds.

We also set out how we will tackle the effects of climate change – still perhaps the most serious long-term risk to the environment given higher land and sea temperatures, rising sea levels, extreme weather patterns and ocean acidification, which harms marine species.

The UK Government has a role in protecting and improving the environment both at home and abroad. We will show leadership on conservation, climate change, land use, sustainable global food supplies and marine health.

We will champion sustainable development, lead in environmental science, innovate to achieve clean growth and increase resource efficiency to provide benefits to both our environment and economy, and keep our pledge to hand over our planet to the next generation in a better condition than when we inherited it.

We will also set gold standards in protecting and growing natural capital – leading the world in using this approach as a tool in decision-making. We will take into account the often hidden additional benefits in every aspect of the environment for national wellbeing, health and economic prosperity, with scientific and economic evidence to the fore.

Since the UK Government is responsible for a number of policies and programmes which affect sectors across the UK and internationally, some aspects of the Plan will apply to the UK as a whole. In other areas where environmental policy is devolved and responsibility rests with the Scottish Government, Welsh Government and Northern Ireland Executive, the proposals in this Plan apply to England only.

More broadly, we will work with the Devolved Administrations as we leave the EU to uphold environmental standards and go further to protect our shared natural heritage. We will continue to work with the Devolved Administrations on areas where common frameworks will need to be retained in the future. This Plan does not pre-empt these discussions.

Our 25-year goals

By adopting this Plan we will achieve:

1. Clean air.
2. Clean and plentiful water.
3. Thriving plants and wildlife.
4. A reduced risk of harm from environmental hazards such as flooding and drought.
5. Using resources from nature more sustainably and efficiently.
6. Enhanced beauty, heritage and engagement with the natural environment.

In addition, we will manage pressures on the environment by:

7. Mitigating and adapting to climate change.
8. Minimising waste.
9. Managing exposure to chemicals.
10. Enhancing biosecurity.

Our policies

We will take action on a number of fronts, looking to join up policies in a way that maximises benefits and value for money. We have identified six key areas around which action will be focused. These are:

- Using and managing land sustainably (chapter 1).
- Recovering nature and enhancing the beauty of landscapes (chapter 2).
- Connecting people with the environment to improve health and wellbeing (chapter 3).
- Increasing resource efficiency, and reducing pollution and waste (chapter 4).
- Securing clean, productive and biologically diverse seas and oceans (chapter 5).
- Protecting and improving the global environment (chapter 6).

Putting the Plan into practice

This Plan is a living blueprint for the environment covering the next quarter of a century. It is an ambitious project, made even more so by our use of a natural capital approach, a world first.

As recommended by the Natural Capital Committee, making the vision of a healthier environment a reality requires solid foundations: comprehensive, reliable data; strong governance and accountability; a robust delivery framework, and everyone to play a role.

Leaving the EU presents a unique opportunity to set in motion the behavioural and institutional changes necessary to build sustainable, enduring growth as well as an improvement in our wellbeing. Critical to delivering the outcomes we want to see is an effective governance structure underpinned by environmental principles – on which we will consult early in 2018. We will develop a set of metrics to assess progress towards our 25 year goals.

We will report on progress annually and refresh the Plan periodically to make sure that our actions continue to target the right improvements and make a real difference. Although the Plan is government-led, everyone in society can play their part in improving the natural world, recognising its full value to all.

The Plan sits alongside two other important government strategies. The Industrial Strategy sets out how we will boost productivity across the UK through five foundations – ideas, people, infrastructure, business, environment, and places. Clean Growth is one of the four Grand Challenges laid out in the strategy that will put the UK at the forefront of industries of the future, ensuring that it takes advantage of transformational global trends.

The Clean Growth Strategy sets out the UK's reaffirmed ambition to promote the ambitious economic and environmental policies to mitigate climate change and deliver clean, green growth.



Durdle Door, Dorset

A summary of our policies

Chapter 1: Using and managing land sustainably

1. **Embedding an 'environmental net gain' principle for development, including housing and infrastructure**
2. **Improving how we manage and incentivise land management**
   i. Designing and delivering a new environmental land management system
   ii. Introducing new farming rules for water
   iii. Working with farmers to use fertilisers efficiently
   iv. Protecting crops while reducing the environmental impact of pesticides
3. **Improving soil health and restoring and protecting our peatlands**
   i. Developing better information on soil health
   ii. Restoring vulnerable peatlands and ending peat use in horticultural products by 2030.
4. **Focusing on woodland to maximise its many benefits**
   i. Supporting the development of a new Northern Forest
   ii. Supporting larger scale woodland creation
   iii. Appointing a national Tree Champion
5. **Reducing risks from flooding and coastal erosion**
   i. Expanding the use of natural flood management solutions
   ii. Putting in place more sustainable drainage systems
   iii. Making 'at-risk' properties more resilient to flooding

Chapter 2: Recovering nature and enhancing the beauty of landscapes

1. **Protecting and recovering nature**
   i. Publishing a strategy for nature
   ii. Developing a Nature Recovery Network
   iii. Providing opportunities for the reintroduction of native species
   iv. Exploring how to give individuals the chance to deliver lasting conservation
   v. Improving biosecurity to protect and conserve nature
2. **Conserving and enhancing natural beauty**
   i. Reviewing National Parks and Areas of Outstanding Natural Beauty

3. **Respecting nature in how we use water**
    i. Reforming our approach to water abstraction
    ii. Increasing water supply and incentivising greater water efficiency and less personal use

Chapter 3: Connecting people with the environment to improve health and wellbeing

1. **Helping people improve their health and wellbeing by using green spaces**
    i. Considering how environmental therapies could be delivered through mental health services
    ii. Promoting health and wellbeing through the natural environment
2. **Encouraging children to be close to nature, in and out of school**
    i. Helping primary schools create nature-friendly grounds
    ii. Supporting more pupil contact with local natural spaces
3. **Greening our towns and cities**
    i. Creating more green infrastructure
    ii. Planting more trees in and around our towns and cities
4. **Making 2019 a Year of Action for the environment**
    i. Helping children and young people from all backgrounds to engage with nature and improve the environment.
    ii. Supporting the 2019 Year of Green Action

Chapter 4: Increasing resource efficiency and reducing pollution and waste

1. **Maximising resource efficiency and minimising environmental impacts at end of life.**
    i. Achieving zero avoidable plastic waste by the end of 2042
    ii. Reducing food supply chain emissions and waste
    iii. Reducing litter and littering
    iv. Improving management of residual waste
    v. Cracking down on fly-tippers and waste criminals
    vi. Reducing the impact of wastewater
2. **Reducing pollution**
    i. Publishing a Clean Air Strategy
    ii. Curbing emissions from combustion plants and generators
    iii. Publishing a Chemicals Strategy

iv. Minimising the risk of chemical contamination in our water

v. Ensuring we continue to maintain clean recreational waters and warning about temporary pollution

Chapter 5: Securing clean, healthy, productive and biologically diverse seas and oceans

1. **Introducing a sustainable fisheries policy as we leave the Common Fisheries Policy**
2. **Achieving good environmental status in our seas while allowing marine industries to thrive**

Chapter 6: Protecting and improving our global environment

1. **Providing international leadership and leading by example**
   i. Tackling climate change
   ii. Protecting and improving international biodiversity
2. **Helping developing nations protect and improve the environment**
   i. Providing assistance and supporting disaster planning
   ii. Supporting and protecting international forests and sustainable global agriculture
3. **Leaving a lighter footprint on the global environment**
   i. Enhancing sustainability
   ii. Protecting and managing risks from hazards
   iii. Supporting zero-deforestation supply chains

Putting the Plan into practice

- Consulting on setting up a new independent body to hold government to account and a new set of environmental principles to underpin policy-making.
- Developing a set of metrics to assess progress towards our 25-year goals.
- Refreshing the 25 Year Environment Plan regularly to ensure that collectively we are focusing on the right priorities, using the latest evidence, and delivering better value for money.
- Strengthening leadership and delivery through better local planning, more effective partnerships and learning from our four pioneer projects.
- Establishing a new green business council and exploring the potential for a natural environment impact fund.
- Work closely with a large range of stakeholders over the coming year to identify their contribution to the goals set out in this Plan.

Natural England Commissioned Report NECR260

Digital video aerial surveys of red-throated diver in the Outer Thames Estuary Special Protection Area 2018

First published 17 May 2019

www.gov.uk/natural-england

Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Background

The Outer Thames Estuary SPA was classified in 2010 in recognition of its importance to wintering red-throated diver *Gavia stellata* which is listed in Annex I of the Birds Directive. Wintering red-throated divers occur throughout the Outer Thames Estuary SPA and use the SPA in numbers of international importance. The peak mean population at classification in 2010 was estimated to be 6,466 individuals. This figure was derived from visual aerial surveys between 1989 and 2006/07.

Two aerial surveys of the Outer Thames Estuary SPA were conducted during January and February 2013. These surveys used high resolution digital still images and produced a peak population estimate of 14,161 individuals, more than twice the classified population.

In late 2017, the Outer Thames Estuary SPA was re-classified with additional qualifying features (breeding common tern *Sterna hirundo* and little tern *Sternula albifrons*) and with amended boundaries that include additional areas of importance to these new qualifying features. This report describes the results of two further aerial surveys of the now enlarged Outer Thames Estuary SPA, conducted in early 2018, this time using high resolution digital video imagery.

The primary objective of the project was to determine an up-to-date population estimate of red-throated divers and an improved understanding of the distribution of birds within the now enlarged Outer Thames Estuary SPA. The intention is that the population estimates of red-throated divers will, if it is considered appropriate to do so, be used in conjunction with the results of the surveys in 2013 to generate a revised baseline population for the site and to amend the target for the population abundance attribute for this species within the Conservation Advice package for the SPA.

The population abundance estimate will also provide a comparison for future site monitoring to report on the condition of the site as part of a rolling monitoring programme. This will help Natural England to fulfil existing and anticipated reporting requirements under the Habitats Regulations, the Marine Strategy Framework Directive and the Marine and Coastal Access Act 2009.

A secondary objective of the project was to generate population estimates for, and an understanding of the distribution of, all other species of birds and marine mammals encountered during these winter surveys of the Outer Thames Estuary SPA.

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HiDef AERIAL SURVEYING LIMITED

Bio Consult SH

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Executive Summary

1 HiDef Aerial Surveying Limited ('HiDef') were contracted by Natural England to undertake digital aerial surveys of the Outer Thames Estuary SPA with the core objective being to ascertain numbers of red-throated diver *Gavia stellata*. Recording of secondary species was to take place but was not deemed the priority of the project.

2 This report provides information on the methods undertaken through the survey flights on two dates in February 2018 and the collation of data, including object and species identification, and post survey analysis.

3 Flights were undertaken on 4 and 17 February 2018, with two Diamond Aviation ('DA') 42 Twin Star aircraft flying sorties on each date. Results for these surveys are presented in this report alongside comparisons with previous surveys.

4 When this contract was commissioned (autumn 2017) a decision to extend the existing Outer Thames Estuary SPA to include an additional c.95km² of sea, adjacent to its existing southern boundary off the Essex coast, with a revised pSPA boundary was pending. This was approved during the course of the contract. Therefore, two sets of numbers are presented at key points in the report to describe numbers within the new, extended SPA boundary and within the smaller boundary of the 'original' SPA. The latter allows easier comparisons with previous surveys that covered only the original SPA.

5 The new pSPA enlarges the existing Outer Thames Estuary SPA (classified solely for non-breeding red-throated divers) to include three new areas identified as being of importance for foraging terns *Sterna sp*. breeding at other (already classified) SPAs on shore; these are parts of the Rivers Yare and Bure, a small riverine section at Minsmere, and both estuarine and marine areas around Foulness. The pSPA therefore comprises areas for foraging breeding seabirds and non-breeding waterbirds. The feature of the existing SPA is retained, and new qualifying features are added based on a review of up-to-date bird abundance information. The total area of the Outer Thames Estuary pSPA is approx. 391,910 ha (392km²).

6 This report provides a new population estimate for the original Outer Thames Estuary SPA for red-throated diver of 21,997. The estimated total for the new, enlarged SPA (formerly the pSPA) is 22,280. Both these figures are approximately 3.5 times greater than the notified population of the original SPA (6,466 individuals).

7 This increase in the recorded number of red-throated divers, coupled with the numbers of individuals of other species recorded has resulted in a peak waterbird population estimate across the original SPA of 41,918 and across the newly enlarged SPA of 46,056 individuals. This suggests that on occasion the SPA supports an assemblage of wintering waterbirds well in excess of the size (20,000 individuals) required to consider this to be a qualifying feature of the SPA.

8 Red-throated divers appear to use the additional area now included within the larger revised SPA boundary at low densities and at high tide when mudflats are covered.

9 Anthropogenic effects on the distribution of several species were detected, notably strong displacement of red-throated diver was noted from areas within offshore windfarms and from areas of shipping activity. This lead to higher densities of birds at points furthest from anthropogenic activity.

1 Introduction

1 The Outer Thames Estuary Special Protection Area ('SPA') was originally designated for wintering red-throated diver *Gavia stellata* which is listed in Annex I of the EU Birds Directive. The wintering population of red-throated divers in Great Britain is estimated to be 17,116 individuals (O'Brien and others 2008), representing between 10% and 19% (depending on the areas included) of the NW European non-breeding population. The Great Britain population estimate is derived from shore-based observations together with more specific aerial and boat surveys. Surveys from boats and aircraft have been responsible for identifying much larger numbers wintering in British coastal waters than previously known and the Great Britain population estimate is still considered to be an underestimate (O'Brien and others 2008).

2 Wintering red-throated divers occur throughout the Outer Thames Estuary SPA and use the SPA in numbers of international importance. The peak mean estimate of the population at designation in 2010 was estimated at 6,466 individuals, 38% of the GB population (Natural England & JNCC 2010). This figure was derived from visual aerial survey work between 1989 and 2006/07. Re-surveys of the entire SPA in early 2013, using digital aerial imagery produced a peak population estimate of 14,161 (APEM 2013; Goodship and others 2015)

3 In 2016, a Departmental Brief was the subject of public consultation on a proposal to add new species to the list of qualifying features of the Outer Thames Estuary SPA and to amend the near-shore boundaries of the site to increase its size to accommodate areas of importance to these additional species of terns (Natural England & JNCC 2015). The Departmental Brief (Natural England & JNCC 2015) did not however, due to a lack of sufficient new evidence, propose any amendment to the seaward boundary of the site or any increase in the size of the population of red-throated divers supported, due to its extended area. At the time of project planning (July 2017) the site was, therefore, considered to be a potential Special Protection Area (pSPA) with altered boundaries and additional features. These changes were adopted in October 2017, during the contract period, resulting in a small increase in area to the SPA boundary, but no associated change to the size of the population of red-throated diver protected within the site.

4 HiDef Aerial Surveying Limited ('HiDef') was contracted by Natural England after a competitive tender process in autumn 2017. The core aim of the contract was to undertake one digital photographic/video aerial survey of the areas within the boundary of the Outer Thames Estuary SPA (and within the amended boundaries of the (at that time) Outer Thames Estuary pSPA) (Figure 1). This resulted in the survey area in the southern sector increasing in size from 2209km² to 2304km². The surveys were proposed to be conducted between 1 November 2017 and 28 February 2018 and the survey images analysed to derive design-based estimates of the red-throated diver population within the SPA/pSPA and the populations of any other species of interest captured in the survey images.

5 Written reports were to be provided, including a description of results of analysis to calculate estimates of abundance (and confidence intervals around those) of red-throated divers in the SPA and its sub-sections, and maps of their distribution. It was also agreed

that files of the processed data and GIS shapefiles generated from the data analysis would be provided as part of the contract.

6 There were two core objectives for this contract. These were:

- A: To conduct one high precision digital aerial survey to determine a new, statistically robust, design-based, baseline population estimate for wintering red-throated divers in the Outer Thames Estuary SPA/pSPA.
- B: To produce maps showing each of: i) the transects flown, ii) the raw sightings data and iii) a density surface derived from simple statistical analyses of those data (for example, kriged interpolation, kernel density estimation ('KDE') etc.) depicting the distribution and spatial variation in the density of red-throated divers within the Outer Thames Estuary SPA/pSPA. (Maps should be fully compatible with ESRI ArcGIS 9.3.1)

7 There were also several optional objectives specified in the invitation to tender, of which the following formed part of the final contract:

- C: To conduct a second, high precision digital aerial survey to determine a second, new, statistically robust, design-based baseline population estimate for wintering red-throated divers in the Outer Thames Estuary SPA/pSPA.
- D: As for B but repeated using the data from the optional 2nd survey.
- I: To produce statistically robust, design-based population estimates of other bird and marine mammal species/groups of interest within the Outer Thames Estuary SPA/pSPA on one survey conducted in fulfilment of objective A.
- J: To produce maps showing each of: i) the transects flown, ii) the raw sightings data and iii) the density surface derived from simple statistical analyses of those data (for example kriged interpolation, KDE etc.) depicting the distribution and spatial variation in the density of each other species/group of bird and marine mammal of interest within the Outer Thames Estuary SPA/pSPA on one survey conducted in fulfilment of objective A. (Maps should be fully compatible with ESRI ArcGIS 9.3.1)
- K: As for objective I but repeated using the data from the optional 2nd survey (in fulfilment of objective C).
- L: As for J but repeated using the data from the optional 2nd survey (in fulfilment of objective C).

8 HiDef provided a description at tender stage of how the works would be completed, providing details of the HiDef digital video aerial survey method, a power analysis of potential survey scenarios, a proposed survey design based on these scenarios, a description of the proposed analysis methods and reporting outline.

9 The need to access parts of the survey area to which access is constrained by Ministry of Defence activities, restricted planned flights to weekends. Inclement weather on all weekends during January 2018 precluded planned surveys in that month. Thus, two separate surveys were undertaken, in agreement with Natural England, on 4 and 17

February 2018, following the survey design shown in Figure 2. Results and analysis for these surveys are presented in this report.

2 Methods

2.1 Survey flights

10 A series of strip transects was flown on two dates: 4 and 17 February 2018, following protocols agreed with Natural England in November 2017.

11 HiDef designed the survey methodology to provide information suitable to support an assessment of the abundance and distribution of red-throated diver, other seabird species and marine mammals in the designated SPA as well as collecting automated (vessel) identification system ('AIS') data, primarily within the southern area of the survey.

12 HiDef designed a survey that placed transects at 3.3 km apart across the southern survey area in which red-throated diver densities are on average greatest, and at 5km apart across the two northern survey areas in which diver densities tend to be less (Figure 1 and Figure 2). The survey design consisted of 42 transects and achieved 13% coverage of the site overall; 15% in the high density southern stratum and 10% in the low-density stratum in the northern sector.

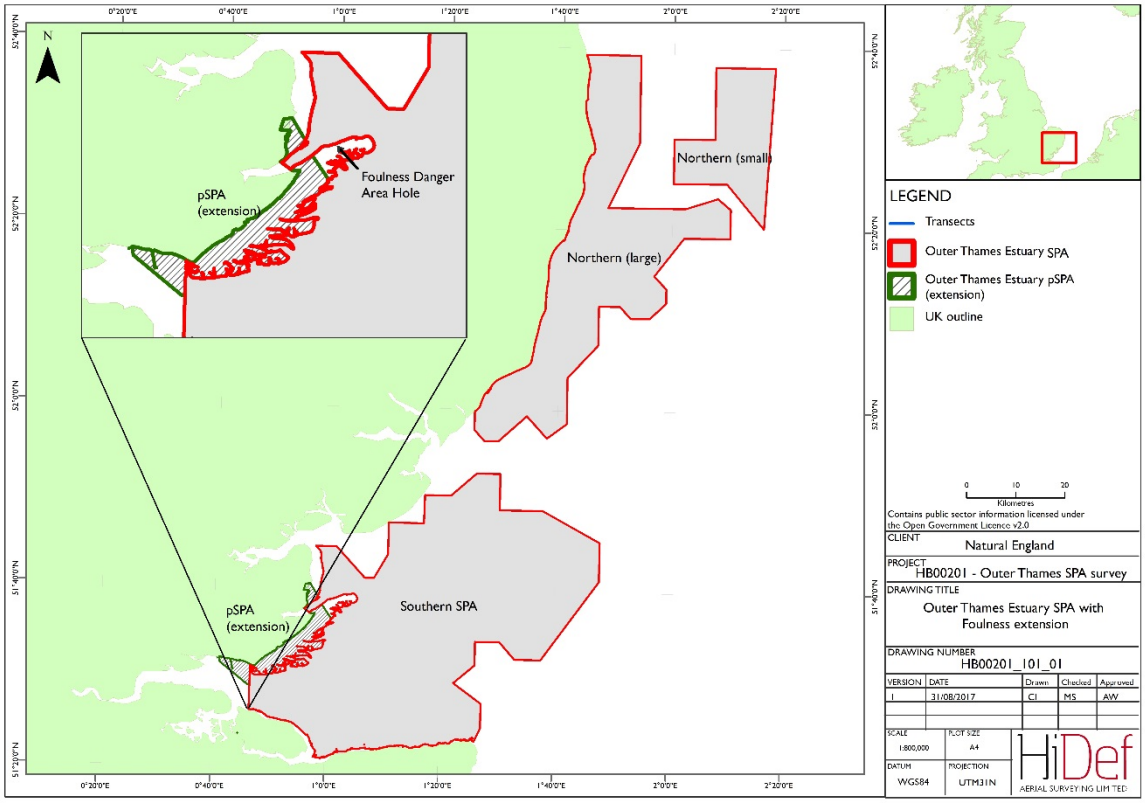


Figure 1 Survey areas for the Outer Thames Estuary SPA with (inset) 'Foulness pSPA extension' and the Foulness Danger Area Hole*

*The landward boundary of the original Outer Thames Estuary SPA abutted that of the Foulness SPA and so excluded the intertidal flats within that SPA. The boundary of the Outer Thames Estuary pSPA was amended to include all areas considered to be of greatest importance to foraging common terns originating from the Foulness SPA; that is all sea areas below mean high water mark within a certain distance from the colony (Natural England and JNCC 2015). Although covering most of the Foulness flats, the limit to this area of importance to the terns did not extend

across the north-easternmost parts of the Foulness SPA mudfalts and so this area remains excluded from the pSPA boundary – so creating a 'hole'. This hole happens to lie within the (military) aviation danger areas D138 and D138C.

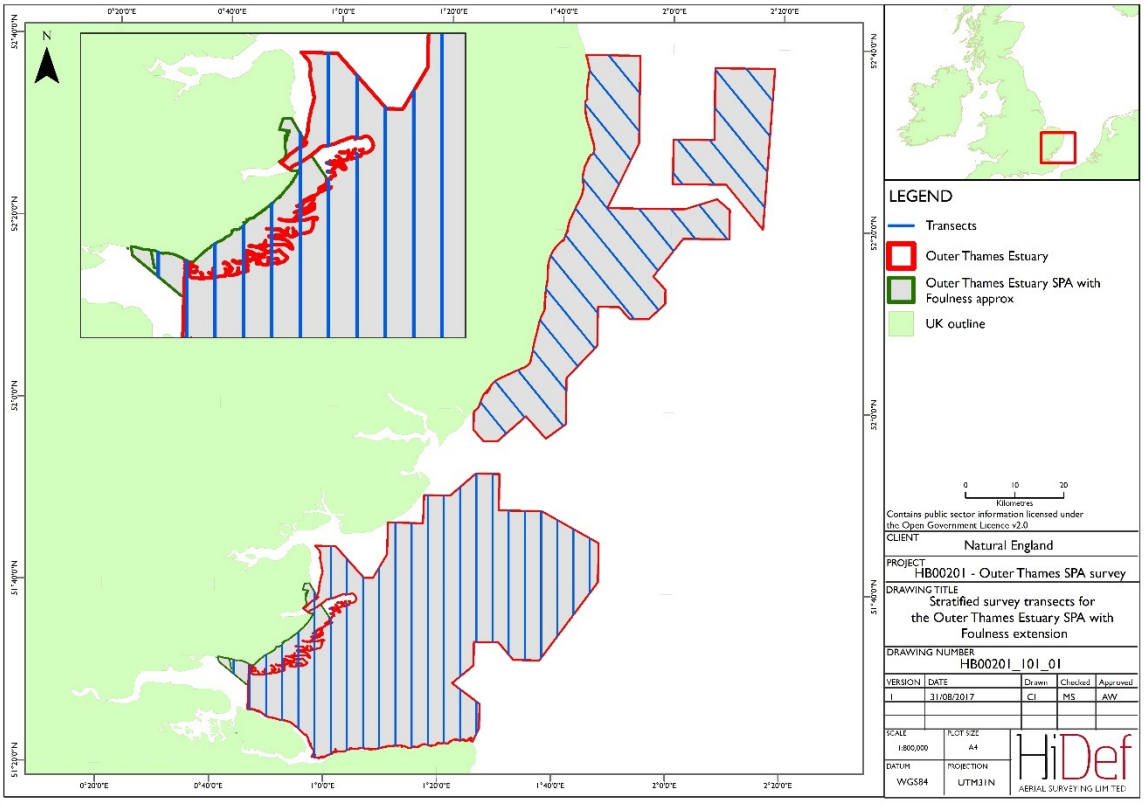


Figure 2 Survey design showing stratified survey transects for the Outer Thames Estuary SPA with (inset) 'Foulness pSPA extension'

13 As part of the survey design process, using survey data gathered under contract to Natural England by APEM Ltd in 2013 (APEM 2013), and provided to all tenderers by Natural England, a power analysis of different survey scenarios was conducted. HiDef ran power analysis of potential survey scenarios using a design-based analysis of the survey data provided. This required research to replicate the analysis carried out in APEM (2013) owing to insufficient detail of methods in this report and in the survey meta-data. In this we clipped all survey data to within the existing SPA boundary, assumed that all images averaged 201m x 269m in dimension, leaving 63 transects within the boundary (64 were flown in both surveys). Distance 7.1 was used to generate abundance estimates, measures of coefficient of variance ('CV') and bootstrapped confidence intervals, in which a uniform model was used (with 0 adjustment terms), assumed poisson distribution and right truncated at 0.13434 km (Table 1). Transect length was adjusted by combining whole or part transects of data to simulate increasing the strip width. This method was independently peer reviewed by DMP Statistical Solutions.

Table 1 Results of power analysis of different potential design-based survey scenarios for the OTE SPA

Scenario	Number transects	Abundance	L95% CI	U95% CI	CV (%)
January non-stratified survey, 269m total transect width	63	11,433	9663	13296	7.82
January survey, non-stratified survey, 500m total transect width	32	11,476	9622	13543	9.93
February non-stratified survey, 269m total transect width	63	14,737	9062	21,213	21.36
February survey, non-stratified survey, 500m total transect width	32	14,710	8072	21,037	22.37
February survey, 500m total transect width, 15 transects in high density stratum and 17 in low density stratum.	32	14,279	9423	18,708	16.91
As above but 19 transects in high density stratum (5 random iterations)	36	14,174	9682	18,299	15.38
As above but 24 transects in high density stratum (10 random iterations)	40	14,763	10,920	18,335	12.94

14 HiDef concluded that there was a discrepancy in the CVs between the new calculations and those published in APEM (2013); HiDef assume that the CV scores for January applied to February and vice versa. We concluded that using fewer, more widely spaced transects of about double the width of those used in 2013 would result in a slight increase in CV. Stratification into two regions would improve the precision but increasing the effort by 50% in the high diver density stratum based on February data would give sufficient coverage to be reasonably certain of achieving about 13% CV, even during the February survey when uncertainty in the abundance estimates was greatest.

15 Additional improvements in precision could potentially be achieved by better orientation of the transects to cross perpendicular to the diver density gradient (not the case in 2013) and by attempting to ensure that all surveys were completed in a single day (the survey was spread over two days in January 2013 and over several days in February 2013), to reduce the amount of variance caused by bird movement between zones.

16 As a result of this investigation HiDef undertook a transect-based survey design in which strip transects were placed approximately perpendicular to the depth contours along the coast. Such a design increases the chance that each transect samples a similar range of habitats (primarily relating to water depth) and helps to reduce the difference in bird and mammal abundance estimates between transects.

17 Surveys were undertaken using two aircraft each equipped with four (4) HiDef Gen II cameras with sensors set to a resolution of 2 cm Ground Sample Distance ('GSD'). Each camera sampled a strip of 125 m width, separated from the next camera by ~20 m, thus providing a combined sampled width of 500 m within a 575 m overall strip (Figure 3). Data from all four cameras was reviewed.

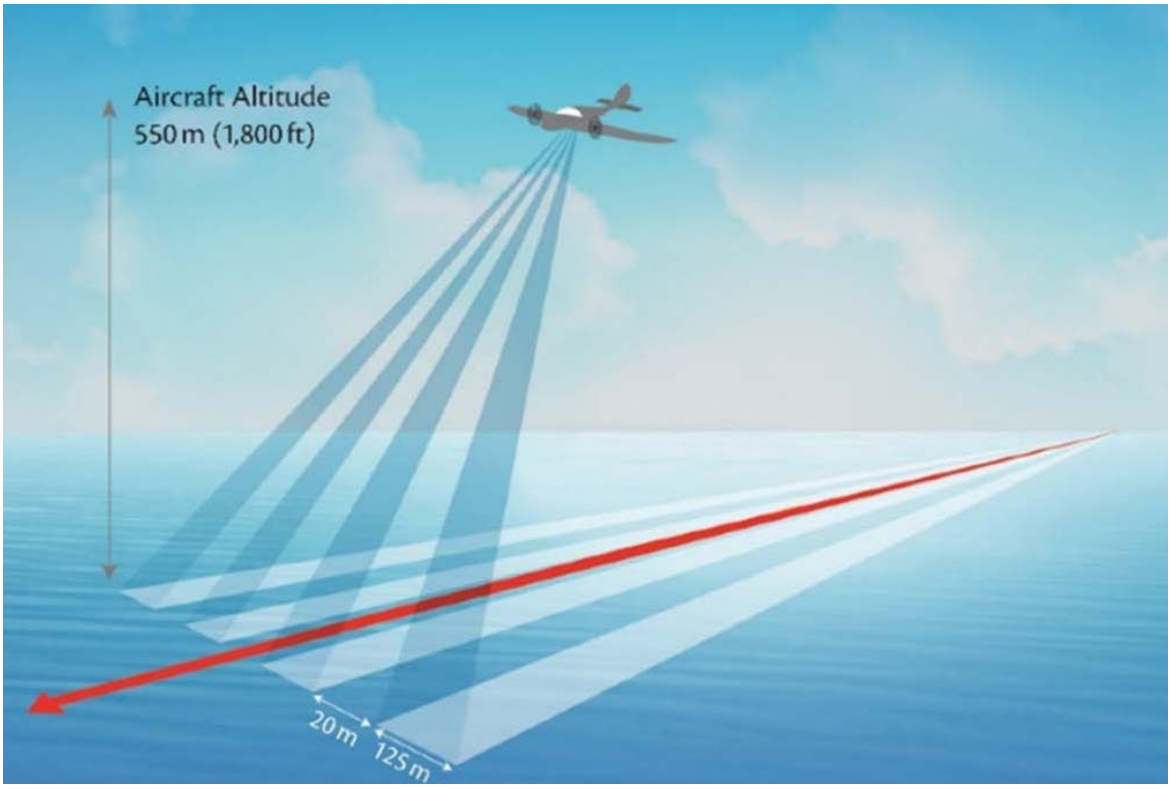


Figure 3 Illustration of HiDef video camera system in operation with cameras pointed slightly forwards and showing strip width of imagery recorded

18 The surveys were flown using a Diamond DA42 aircraft flying along the transect pattern shown in Figure 2 at a height of approximately 550 m above sea level ('ASL') (~1800 ft). Flying at this height ensures that there is no risk of flushing those species which have been proven to be easily disturbed by aircraft noise and presence, such as red-throated diver and scoter.

19 It is also a height that allows good clearance of offshore structures such as wind turbines, which require a minimum 150 m clearance from aircraft at all times. This is a stipulation of the European Aviation Safety Agency (EASA).

20 Position data for the aircraft was captured from a Garmin GPSMap 296 receiver with differential GPS enabled to give 1m for the positions and recording updates in location at 1 second intervals for later matching to bird and marine mammal observations.

21 At the early planning stage, it was identified that the survey design would take more than one day to complete due to the area of coverage. To minimise the effect of movement of red-throated divers between transects and different survey sectors, HiDef endeavoured successfully to complete all surveys in a single day by deploying two aircraft on the same day. This allowed a more effective 'snapshot' of the population to be achieved. Both aircraft commenced surveys from a central position, moving further apart as the survey progressed – one aircraft moving overall northwards, the other to the south. The phenomenon of red-throated diver movements in the Outer Thames was highlighted by Webb and others (2009) and could potentially cause significant bias to any abundance estimates from surveys that do not minimise this effect.

22 This method differed from the two high resolution digital stills aerial surveys of the Outer Thames Estuary SPA that were conducted during multiple days in January and February 2013 (APEM 2013). Each survey then was flown on a series of transects separated by 1.8 km, collecting abutting 3 cm resolution imagery. Average coverage was 15%. Population estimates of red-throated divers were calculated using two methods: 1) a design-based method and 2) a model-based method using a Generalised Additive Modelling ('GAM') framework.

2.2 Data Review and Object Detection

23 Data were viewed by trained aerial data reviewers who marked any objects in the footage as requiring further analysis, as well as determining which objects were birds, marine megafauna (defined within this report as cetaceans, pinnipeds or other large, non-avian marine fauna) or anthropogenic objects such as ships or buoys.

24 As part of HiDef's quality assurance ('QA') process, an additional 'blind' review of 20% of the raw data was carried out and the results compared with those of the original review. If 90% agreement is not attained during the QA process, then corrective action is initiated: the remaining data set is reviewed and where appropriate, the failed reviewer's data discarded and all the data re-reviewed. In addition, additional training is then given to the reviewer to improve performance.

25 As part of the video review process objects are only recorded where it reaches a reference line (known as 'the red line') which defines the true transect width of 125m for each camera. Due to each object being recorded in multiple video frames it is essential to ensure that the object is only recorded once, and not multiple times. As such the 'red line' system is introduced to ensure accurate data recording. By excluding objects that do not cross the red line, biases to abundance estimates caused by flux (movement of objects in the video footage relative to the aircraft, such as 'wing wobble') are eliminated.

2.3 Object Identification

26 Images marked as requiring further analysis, that is those in which objects had been detected, were reviewed by specialist ornithologists[1] for identification to the lowest taxonomic level possible and for assessment of the approximate age and the sex of each animal, as well as any behaviour traits visible from the imagery.

27 At least 20% of all objects were subjected to an external QA process. If 90% agreement is not attained then corrective action is initiated: if appropriate, the failed reviewer's data is discarded and the data re-reviewed. Any disputed identifications are passed to a third-party expert ornithologist for a final decision[1].

28 Each animal was assigned to at least a species group (genus/family), and where possible each was also assigned a species identification with confidence levels of 'Possible', 'Probable' or 'Definite'. Any animals that could not be identified to species level were assigned to a category 'No ID' in the species column.

29 The aims of identification are four-fold:

- To identify objects detected by the reviewers and classify as many to species level as possible;
- To discriminate between animate and inanimate objects, and where apparent to detect new objects;

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1 HiDef currently employs three (3) of the ten (10) current members of the British Birds Rarities Committee ('BBRC') as expert ornithologists

- To record additional information where possible, such as behaviour, age and sex;
- To ensure identifications are as accurate as possible through a robust audit process.

30 Confidence in identification is based on qualitative rather than quantitative factors based on a uniquely skilled aerial imagery identification team:

- ‘Definite’: as certain as reasonably possible;
- ‘Probable’: very likely to be this species or species group;
- ‘Possible’: more likely to be this species or species group than anything else.

31 Quantifying percentage certainty is a very subjective matter. For example, if you are 75% certain that a definite ‘large auk’ is a guillemot, it does not follow that there is a 25% chance that it is a razorbill. What it actually takes to reach levels of certainty are the prescriptive number of identification features that an image shows and how clear and robust these features show, that is colour, tone, size, shape and jizz.

32 Objects were identified to species where possible, or alternatively fall within wider species groups (Table 2). Most species groups are self-explanatory. The exception however is that on occasion auks and divers, if undertaking behaviour such as diving, can be more problematic to identify.

Table 2 First level species groups used in identification process prior to species categorisation

Species group (No ID)	
Duck species	Gull species
Diver species	Large auk
Fulmar / gull species	Auk species
Cormorant / shag	Auk / small gull
Wader species	Large auk / diver species
Small gull species	Passerine species
Black-backed gull species	Seal species
Large gull species	Seal / small cetacean species

33 Surfacing behaviour was defined as any surfacing behaviour that occurred while the non-avian animal was visible. However, for harbour porpoise surfacing behaviour was also classified as 'snapshot surfacing' if the animal's dorsal fin was above the water in the frame nearest to the horizontal centre line.

34 Additional information was recorded for each bird on their basic behaviour (whether the bird was sitting, loafing on land or other objects or flying; in the latter case the direction of travel was also recorded. More detail was recorded where possible on foraging behaviour, approximate age and sex and any other details of interest.

2.4 Final processing

35 All data were geo-referenced, taking into account the offset from the transect line of the cameras, and compiled into a single output; Geographical Information System ('GIS') files for the Observation and Track data are issued in ArcGIS shapefile format, using UTM31N projection, WGS84 datum.

2.5 Data analysis

2.5.1 Data treatment

36 After basic review and identification had been undertaken, data were processed for estimating abundance and distribution of the key species and species groups. In line with the core objectives the focus remained on red-throated divers.

37 For species groups which include different genera (for example small gull/auk), species level identification is used to assign to species group (for example large auk). Where identification to species level isn't possible, a broader species group category is instead used for that record (for example large auk rather than allocating exactly to guillemot or razorbill).

2.5.2 Abundance Estimates

38 The abundance of each species observed is provided. Both raw data and post identification and audit analysis are presented.

39 Density and population estimates, as well as the upper and lower 95% confidence limits were derived by way of a blocked bootstrapping technique in order to ensure equal transect effort was sampled across each iteration. This was done by using transect ID as the sampling unit with replacement, and then randomly sampling until the total length of the sampled transects equalled approximately the total length of transects surveyed. We calculated the mean and standard deviation of the sampled means, as well as the relative standard error as defined by the standard deviation divided by the mean. Data were processed in the R programming language (version 3.4.3) and code can be provided on request.

40 Distance sampling is a commonly used method of calculating the density and abundance of animals from surveys that considers the problem of detection probability. The key concept of the distance method is that the probability of detecting an animal decreases as the distance from the observer increases (Buckland and others 1993). It isn't generally required for digital aerial data because of the uniform detection rates at all distances from the transect line. However, HiDef used it because it is a well-established method for calculating density and abundance from transect-based data using design-based methods even without employing the distance component of the application.

41 In the case of this aerial survey, we can apply the distance sampling technique because we have a known strip transect width and cannot detect birds outside of that area. Distance allows us to take this issue into account for good estimates of density and abundance. Furthermore, we can bootstrap this output within the distance framework to obtain 95% confidence limits on our estimates.

42 This Distance sampling method was applied to red-throated diver only as this was the focal species of the survey. As the focal species, this approach added certainty and validation to the alternative method used, which is known as 'block bootstrapping' which was applied to all other species.

43 For this analysis, we used Distance version 7.1 (Thomas and others 2010) using the conventional distance sampling analysis engine. Data were stratified by region (that is the large Northern area, the smaller Northern area, and the Southern SPA area), and then by transect. The transect stratum was used to calculate the encounter rates, and the global

density estimate was calculated as the mean, weighted by the transect areas. We applied a uniform – cosine detection function with a fixed two-sided width of 250m (the birds could only be seen within each 250m transect band), and no adjustment terms were applied. Bootstrapping was performed via a non-parametric bootstrap, 1000 times, with the observations being resampled from within each transect.

44 Using the same transect sampling regime to calculate the population estimate of red-throated divers in the original (smaller) SPA and the new, extended SPA boundary resulted in a larger point estimate of the population size in the smaller original SPA boundary than in the revised, larger boundary. While this result is counter-intuitive, it is not unusual for this to occur when the two sites are very close to each other in size, especially given that the confidence limits for both estimates overlap substantially. To provide a more useful estimate of the population size of red-throated divers within the additional sea area now included within the revised, larger SPA and, for comparison with previous surveys of the old SPA boundary, we calculated the population size of red-throated divers in the extension area using the output from the kernel density estimation ('KDE') modelling (see below). The population size within each grid cell (polygon) of the KDE surface for red-throated divers was calculated by multiplying the density by the surface area of each cell. Using ArcGIS 10.4, we selected the grid cells that occurred in the extended part of the new SPA boundary and summed the population size of the selected cells. This gives a population estimate but no confidence intervals for this small area.

2.5.3 Density Mapping

45 The density maps have been derived using a Watson-Nadaraya type kernel density estimation ('KDE') technique (Simonoff, 1996). In KDE, a small 'window' function (the kernel) is used to calculate a local density at each point in the study area. To evaluate the density at a given point, the kernel is centred on that point and all the observations within the window are summed to obtain a local count. The total area of the transect(s) intersecting the window is then summed to obtain a local measure of effort. By dividing the local count by the local effort, a local density estimate is obtained. To build a density map, the study area is covered with a fine mesh of study points and the density is calculated at each point in the mesh in turn.

46 Kernel techniques are robust and not as complex as other density estimation techniques because they have few parameters; as a result, they are arguably the easiest density surface technique to reproduce independently. The only variables are the size and shape of the kernel or window function. For these analyses, we have used a Gaussian window function, which has the advantages of being smooth, rotationally symmetric and easy to compute. The shape of the Gaussian is determined by a single width parameter; the selection of this parameter is the only variable in the computation of the density maps.

47 Rather than set the width parameter arbitrarily, we have used a leave-one-out cross validation method. Cross validation estimates the predictive power of a model by removing some of the data from the data set and using the remainder of the data and the model to predict the values for the data that were removed. The closer the predicted values represent the removed data, the better the model performance and the width parameter used in the model.

48 To apply cross validation to the survey area, each transect is subdivided into 1 km long segments. To evaluate a particular choice of kernel width, each segment is removed in turn, the kernel and the remaining data are used to predict the density of the missing segment and the known value subtracted from the prediction to obtain an error score. This process is repeated for every segment and the error scores for all segments are squared and summed to give a total performance score for that particular choice of kernel width. The kernel width is then varied and the process repeated; if the new score is lower than the old, the new kernel width is a better choice than the previous value. An exhaustive search over all kernel widths is then used to identify the best global choice. The result of the process is a smooth density estimate which has been derived without any manual parameter selection. The whole process is repeated from scratch for each map, as different kernel sizes are appropriate for different species.

49 It should be noted that several of the KDE maps are effectively flat. These correspond to distributions where the density surface as obtained from a small local kernel was not effective at predicting missing data; this can happen with evenly distributed birds but can also happen for very sparse distributions. In the case of sparse distributions, the 'flat' map does not necessarily mean that the true underlying distribution is 'flat'; it could mean that the data doesn't contain enough evidence to determine what the underlying distribution is. It is therefore useful to refer back to the population estimates for the corresponding map when looking at these 'flat' densities; we have also overlaid the relevant raw observations as dots to help with interpretation of the maps. In extreme cases, the density maps were not included in the results section, and the data presented simply as dot maps.

3 Additional survey parameters

3.1 AIS data collection

50 HiDef installed an automated identification system ('AIS') receiver in one aircraft, flying over the southern sector, for both surveys, which provided detailed real-time information on the location of different ship types. This is considered to offer superior detail on ship activity that may cause disturbance to red-throated divers than might be obtained from simple monthly summaries of shipping activity. Tracking outputs are presented here (Figure 4 and Figure 5) to illustrate the type of data collected and the routes followed by vessels. The range of the AIS receiver was less than hoped, in relation to the northern area, but good data were received from the high-density stratum in the south. In a further analysis, the output from Figure 4 was used to define the location of shipping lanes in a simple GIS analysis, in which distance of outputs of raw observations and modelled KDE outputs of red-throated diver distribution were measured from the line representing the shipping lanes.

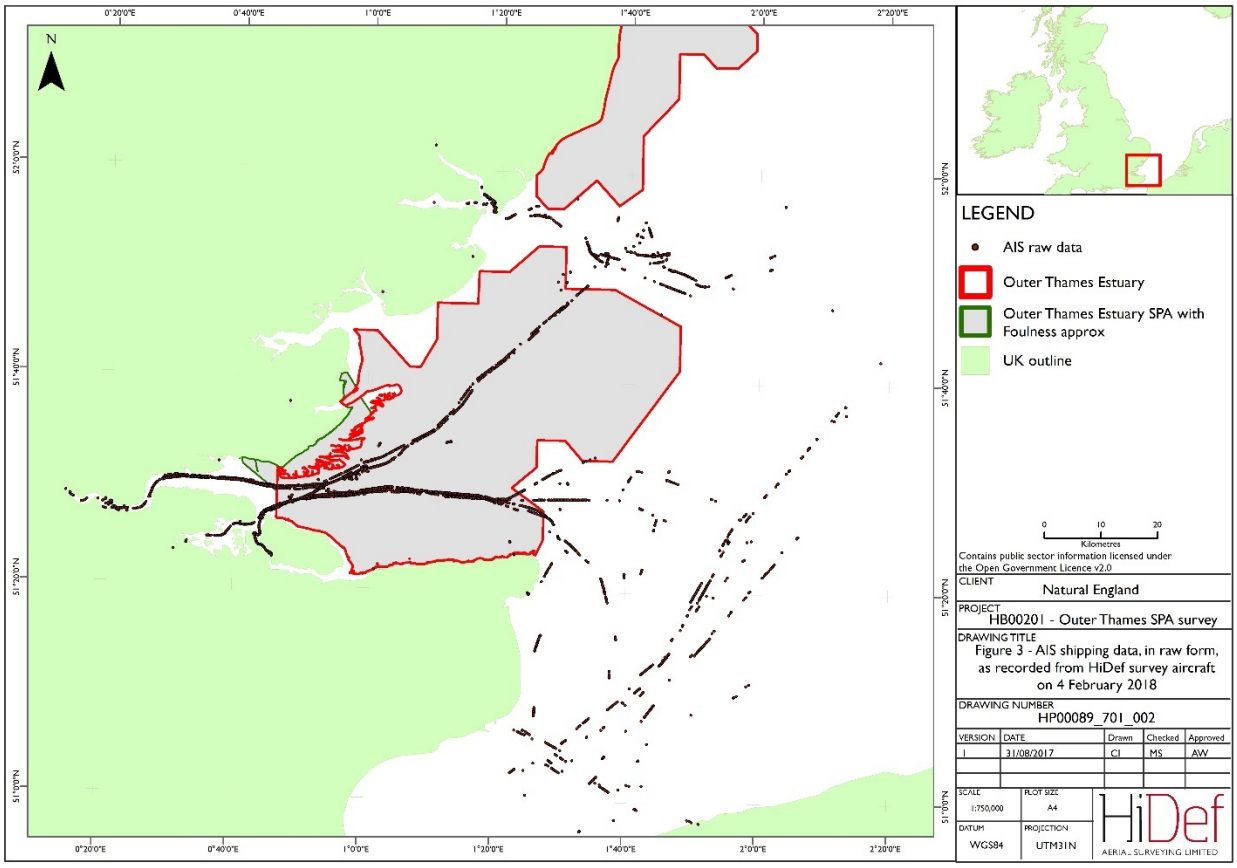


Figure 4 AIS shipping data, in raw form, as recorded from HiDef survey aircraft on 4 February 2018

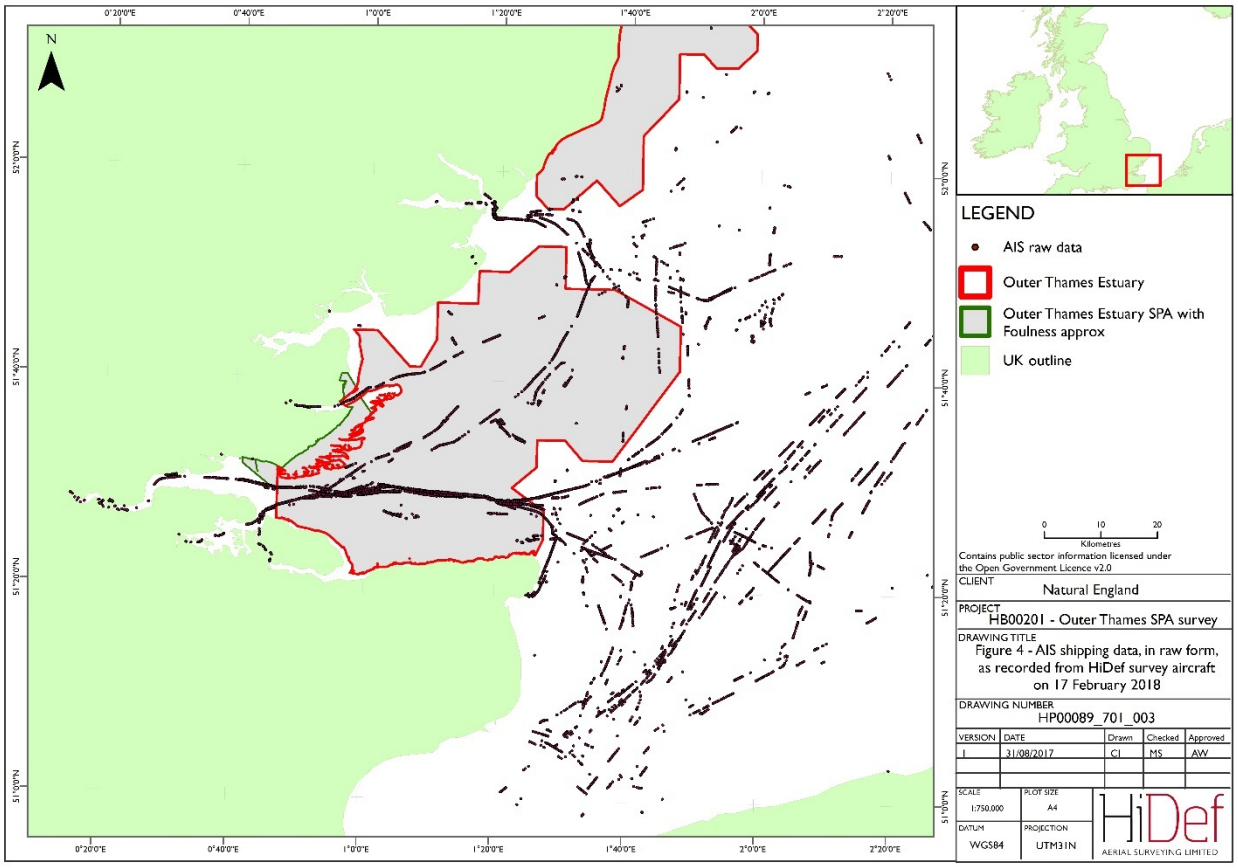


Figure 5 AIS shipping data, in raw form, as recorded from HiDef survey aircraft on 17 February 2018

3.2 Flight restrictions

51 HiDef noted the presence of (military) aviation danger areas D138 and D138C located around Foulness (firing ranges) which are often active up to 6000 ft. ASL. While military activity is broadcast through notices to airmen ('NOTAMs'), military ranges are usually active throughout daylight hours. HiDef staff negotiated access to this area by prior contact with the range controllers at Shoeburyness range and with the contractor Quinetiq. Potential 'cold' periods for the range, when it would be possible to penetrate the danger areas safely were identified. These all related to weekends, and no times during weekdays were offered despite repeated requests. Unfortunately, no suitable weather windows in January 2018 corresponded with these restrictions but fortunately, they did so on 4 and 17 February 2018 when surveys took place.

52 Aircraft flight height was maintained at c.550 m to ensure full legal clearance over man-made structures in the area such as wind turbines. These must be overflown with 150 m clearance as a minimum.

4 Project Management

53 Post tender acceptance, an initial meeting was held by telephone between Richard Caldow of Natural England and Martin Scott and Andy Webb of HiDef.

54 HiDef agreed to tailor flights to around high tide in the Foulness area, where possible, or to at least times when mud flats would not be fully exposed, increasing the possibility of usage by divers. This was done as well as possible within the constraints imposed by only being able to conduct flights in this area at weekends.

55 Temporal spacing of flights was agreed. Ideally this was to be one flight in late January and the other in mid-February, but weather and military restriction issues did not allow this. Both flights were eventually flown, 13 days apart, in February with prior agreement from Natural England.

56 A press statement was agreed by both parties and published on the HiDef website (http://hidef.bioconsult-sh.de/en/news-archive/natural-england-chooses-hidef-for-outer-thames-surveys/).

57 Ongoing dialogue continued between HiDef and Natural England during the contract period to ensure smooth delivery first of a draft report, then during fuller data analysis and reporting.

5 Results

5.1 Survey results

58 The analysis of data to species level used all levels of identification confidence, with the overall identification rate of birds and non-avian animals to species level for the two surveys shown in Table 3.

Table 3 Survey identification rates at the Outer Thames in February 2018

Survey date	ID rate (%)
4 February 2018	93.34%
17 February 2018	91.71%
Average	**92.53%**

59 The total number of objects detected in each survey flight, as well as numbers of species and species group are presented in Table 4 to Table 5.

60 Red-throated diver was the principal species of interest in this survey programme. Some 1333 were recorded on 4 February, while 2917 were recorded in total on 17 February. These results were attained by the aircraft following the same transect routes in each survey. Note that the identification of no other diver species was confirmed during these surveys.

5.2 Abundance estimates

61 The density, total estimated population, upper and lower 95% CI, standard deviation and CV for each species and species group have been calculated using strip transect analysis and are presented in Table 6 to Table 29. Data have been presented for the entire original SPA (Table 6, Table 7, Table 10 and Table 11), the entire new pSPA (Table 8, Table 9, Table 12 and Table 13), the southern part of the original SPA (Table 14, Table 15, Table 22 and Table 23), the enlarged southern part of the new pSPA (Table 16, Table 17, Table 24 and Table 25), and the larger (Table 18, Table 19, Table 26 and Table 27) and smaller (Table 20, Table 21, Table 28 and Table 29) northern parts of the SPA/pSPA (which are effectively unchanged). Highlights for the species observed are listed in this section. A comparison of the 2013 and 2018 data is provided in Table 32.

62 Densities for all birds were recorded for each of the surveys and had relatively high densities. During Survey 1 on 4 February, density estimates for all birds in all areas of the original SPA were recorded at 4.80 birds/km² and at 4.86 birds/km² in the newly enlarged SPA. This equated to 18,260 birds (± 95% CI 14,273 – 22,410) in the original SPA and 19,071 birds (± 95% CI 14,990 – 23,411) in the newly enlarged SPA. During Survey 2 on 17 February, density estimates increased and were recorded at 11.01 birds/km² and 11.73 birds/km² in the original SPA and the newly enlarged SPA respectively. This equated to a population estimate of 41,918 birds (± 95% CI 32,965 – 51,586) in the original SPA and 46,056 birds (± 95% CI 36,150 – 57,008) in the newly enlarged SPA.

63 Low densities of common scoter *Melanitta nigra* were recorded at 0.23 birds/km² and 0.07 birds/km² in Survey 1 and Survey 2 respectively in the southern area (SPA). This equated to 515 birds (± 95% CI 0 – 1480) and 161 birds (± 95% 0 – 466). In the pSPA, density estimates were calculated at 0.22 birds/km² which equated to 513 birds (± 95% CI 0 – 1485) in Survey 1 and 0.07 birds/km² in Survey 2 which equated to 160 birds (± 95% CI 0 – 466). Within the northern areas, common scoter were recorded only in the large area at 0.42 birds/km² in Survey 2 equating to 509 birds (± 95% CI 0 – 1466).

64 One surf scoter *Melanitta perspicillata* was recorded in the southern area during Survey 2.

65 Three red-breasted mergansers *Mergus serrator* were only recorded in the southern area during Survey 2.

66 Red-throated diver was the most abundant species and was recorded in all areas with an increase in density recorded across all areas between Survey 1 and Survey 2. In Survey 1 (SPA southern area), red-throated diver density was calculated at 3.64 birds/km² which equated to 8021 birds (± 95% CI 6421 – 9695) and increased to a peak density of 7.10 birds/km² in Survey 2, equating to 15,671 birds (± 95% CI 10,531 – 21,265). Across the whole of the larger pSPA southern area, including the Foulness extension, density estimates for red-throated divers were somewhat lower at 3.51 birds/km² in Survey 1 and 6.88 birds/km² in Survey 2. This equated to a population estimate of 8161 birds (± 95% CI 6520 – 9841) and 16,002 birds (± 95% CI 10,841 – 21,682) respectively. In the large northern area, red-throated diver density was calculated at 0.62 birds/km² in Survey 1 and 3.77 birds/km² in Survey 2. This equated to 756 birds (± 95% CI 504 – 1019) and 4587 birds (± 95% CI 2499 – 7114) respectively. In the small northern area, the density estimates were calculated at 0.34 birds/km² and 0.53 birds/km² for this species equating to 132 birds (± 95% CI 76 – 196) and 204 birds (± 95% CI 137 – 274) in Survey 1 and Survey 2 respectively. Overall, across the entire original SPA the density estimate for red-throated

divers was 2.66 birds/km² in Survey 1 equating to 10,136 birds (± 95% CI 7763 – 12,626) and 5.78 birds/km² in Survey 2 which equated to 21,997 birds (± 95% CI 15,351 – 29,415) within the SPA. Across the entire pSPA area, the density estimates for the species were calculated to be 2.58 birds/km² in Survey 1 equating to 10,148 birds (± 95% CI 7868 – 12,544) and 5.68 birds/km² in Survey 2 equating to 22,280 birds (± 95% CI 15,611 – 29,784). Thus, while the average density estimates of red-throated divers across the entire pSPA were somewhat lower than in the original SPA boundary, the population estimates within the pSPA are slightly greater than in the original SPA boundary. Only three divers were not identified to species level over both survey dates (Table 3). A summary table for red-throated diver is presented in Table 30.

67 Figure 6 shows how the red-throated diver density changes between the two surveys. It also shows how the density differs between the London Array, Kentish Flats and Gunfleet Sands windfarms compared to the remainder of the site (wind farm areas based on data from Crown Estate March 2018). The marked difference is suggestive of a significant displacement effect. There is an opportunity for further analysis of this effect to be investigated, but this is out with the scope of this contract.

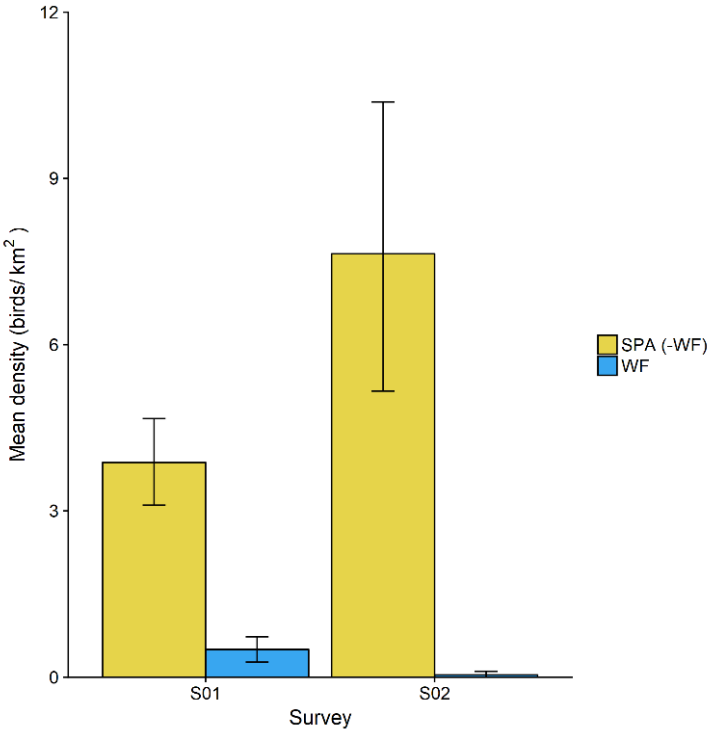


Figure 6 Red-throated diver mean density (birds/km²) within the SPA, excluding the areas within the footprints of wind farms (SPA (-WF)), and within those windfarm footprints (WF) for Survey 1 and Survey 2.

68 In 2013, multiple black-throated diver *Gavia arctica* and great northern diver *Gavia immer* were recorded (APEM, 2013). None were recorded in 2018. Historical surveys show both species to be very rare in the Outer Thames with low single figure counts recorded in survey between 1989 and 2007 (Webb and others 2009). 2018 counts failed to record either of these species, again suggesting they only occur very rarely. This contrasts markedly with the assertion made in the report of the 2013 surveys (APEM 2013) which provided population estimates in the hundreds of each species (Table 32).

69 Fulmar *Fulmarus glacialis* density estimates were calculated at 0.03 birds/km² in the large northern area for both surveys with an estimated population at 31 birds (± 95% CI 0 – 71) in Survey 1 and 41 birds (± 95% CI 0 – 82) in Survey 2. Within the small northern area density estimates were recorded at 0.13 birds/km² equating to 51 birds (± 95% CI 18 – 89) in Survey 1 and decreased to 0.05 birds/km² in Survey 2. This equated to 21 birds (± 95% CI 0 – 53). There were no observations of the species within the southern area on either survey.

70 Gannet *Morus bassanus* were recorded in the southern and the large northern areas in Survey 2 only. Within the southern part of the SPA, density estimates were calculated at 0.19 birds/km² which equated to 429 birds (± 95% 138 – 798) and in the southern part of the pSPA, a density was calculated at 0.18 birds/km² also equating to 429 birds (± 95% CI 139 – 792). In the large northern area, a low density estimate of gannets was obtained at 0.03 birds/km² which equated to 41 birds (± 95% CI 10 – 81).

71 Cormorant *Phalacrocorax carbo* were also only recorded in the southern area and large northern area in either survey. In Survey 1 in the southern area (SPA), density was calculated at 0.14 birds/km² and this increased to 0.49 birds/km² in Survey 2. This equated to 431 birds (± 95% CI 117 –784) and 1092 birds (±95% CI 196 – 2397) respectively. In the southern area pSPA, the species density estimate was calculated at 0.19 birds/km² in Survey 1 which equated to 448 birds (± 95% CI 128 – 819) and in Survey 2, a density was calculated at 0.49 birds/km² equating to 1140 birds (± 95% CI 220 – 2468). Within the large northern area, cormorant density estimates of 0.05 birds/km² were recorded in Survey 1 and increased to 0.25 birds/km² in Survey 2 equating to a population estimates of 62 birds (± 95% CI 21 – 104) in Survey 1 and 257 birds (± 95% CI 59 – 546) in Survey 2.

72 Great crested grebe *Podiceps cristatus* density estimates of 0.14 birds/km² and 0.38 birds/km² in Survey 1 and Survey 2 were recorded in the southern area of the SPA. This equated to 315 birds (± 95% CI 71 – 608) and 836 birds (± 95% CI 392 – 1327). In the southern area of the pSPA, density estimates were calculated at 0.14 birds/km² in Survey 1 and 0.36 birds/km² in Survey 2 which equated to 318 birds (± 95% CI 72 – 622) and 839 birds (± 95% CI 402 – 1316) respectively. In the large northern area, great crested grebes were recorded at densities of 0.01 birds/km² equating to 11 birds (± 95% CI 0 – 30) in Survey 1 and 0.08 birds/km² equating to 103 birds (± 95% CI 30 – 187) in Survey 2.

73 Some 44 sanderling *Calidris alba* were recorded (in flight) in the southern survey area during Survey 2.

74 Kittiwake *Rissa tridactyla* density was calculated at 0.32 birds/km² in the southern area of the SPA for Survey 1 and decreased to 0.10 birds/km² in Survey 2. This equated to 710 birds (± 95% CI 475 – 964) and 211 birds (± 95% CI 134 – 297) respectively. In the southern area of the pSPA, a density was calculated at 0.31 birds/km² and 0.09 birds/km² in Survey 1 and Survey 2 which equated to 718 birds (± 95% CI 485 – 973) and 212 birds (± 95% CI 132 – 300) respectively. Density was calculated at 0.04 birds/km²in Survey 1 and 0.07 birds/km² in Survey 2 within the large northern area. In the small northern area, density estimates for kittiwake were calculated at 0.08 birds/km² in Survey 1 equating to 31 birds (± 95% CI 10 – 52) and 0.08 birds/km² in Survey 2 which equated to 31 birds (± 95% CI 0 – 62).

75 Little gull *Hydrocoloeus minutus* were only recorded in Survey 2 in the southern area and the small northern area. Density was calculated at 0.003 birds/km² equating to 7 birds (±

95% CI 0 – 20) in the southern area of the SPA, and 0.003 birds/km² in the southern area of the pSPA equating to 7 birds (± 95% CI 0 – 20) and 0.03 birds/km² in the small northern area which equated to 11 birds (±95% CI 0 – 29).

76 Black headed gull *Chroicocephalus ridibundus* density estimates in the southern area of the original SPA were calculated at 0.19 birds/km² and 0.39 birds/km² in Survey 1 and Survey 2 equating to 414 birds (± 95% CI 105 – 783) and 864 birds (± 95% CI 192 – 1717) respectively. Within the southern part of the newly enlarged pSPA, density estimates were calculated at 0.34 birds/km² and 0.90 birds/km² in Survey 1 and Survey 2 equating to 790 birds (±95% CI 236 – 1488) and 2083 birds (± 95% CI 472 – 4270) in Survey 2. The species was only recorded in the large northern area during Survey 2 with a density estimate of 0.03 birds/km². Within the small northern area, density estimates were similar with estimates of 0.13 birds/km² and 0.18 birds/km² in Survey 1 and Survey 2 which equated to 52 and 71 birds respectively.

77 Mediterranean gull *Larus melanocephalus* density was calculated at 0.01 birds/km² in the large northern area during Survey 1 equating to 11 birds (± 95% CI 0 – 30). Density was calculated at 0.01 birds/km² in the southern area during Survey 2 which equated to 14 birds (± 95% CI 0 – 27).

78 Common gull *Larus canus* was one of the most abundant species. In the southern area of the SPA, density was calculated at 0.44 birds/km² and 1.34 birds/km² which equated to 975 birds (±95% CI 598 – 1416) in Survey 1 and 2946 birds (± 95% CI 1781 – 4325) in Survey 2. For this species density estimates were calculated at 0.46 birds/km² and 1.39 birds/km² in the pSPA southern area for Survey 1 and Survey 2 respectively. This equated to 1076 birds in Survey 1 and 3239 birds in Survey 2. Within the large northern area, density was calculated at 0.10 birds/km² in Survey 1 which equated to 122 birds (± 95%CI 50 – 210) and 0.42 birds/km² in Survey 2 equating to 511 birds (± 95% CI 257 – 796). In the small northern area, density was calculated at 0.21 birds/km² in Survey 1 and 0.08 birds/km² in Survey 2 equating to 81 birds and 31 birds respectively.

79 Lesser black-backed gull *Larus fuscus* density was calculated at 0.01 birds/km² equating to 27 birds (± 95% CI 7 – 46) in the southern area of the both the original SPA and the newly enlarged pSPA during Survey 1. The species was not recorded in either of the northern areas during Survey 1. In Survey 2, density was calculated at 0.12 birds/km², 0.08 birds/km² and 0.03 birds/km² in the southern, large northern and small northern areas respectively. This equated to population estimates of 263 birds (± 95% CI 165 – 376), 93 birds (± 95% CI 51 – 140) and 11 birds (± 95% CI 0 – 30) respectively.

80 Density estimates for herring gull *Larus argentatus* were calculated at 0.28 birds/km² and 0.44 birds/km² in the southern area of the SPA for Survey 1 and Survey 2. This equated to 624 birds (± 95% CI 163 –1375) in Survey 1 and 968 birds (± 95% CI 691 – 1262) in Survey 2. In the southern area of the pSPA, density estimates were calculated at 0.28 birds/km² and 0.45 birds/km² for Survey 1 and Survey 2 equating to 647 birds and 1047 birds respectively. Within the large northern area, density was calculated at 0.06 birds/km² equating to 72 birds (± 95% CI 10 – 149) in Survey 1 and increased to 0.77 birds/km² equating to 931 birds (± 95% CI 457 – 1504) in Survey 2. Herring gull were only recorded in the small northern area during Survey 2 with a density estimate of 0.05 birds/km² which equated to 21 birds (± 95% CI 0 – 44).

81 Great black-backed gull *Larus marinus* density was calculated at 0.48 birds/km² and 0.19 birds/km² in the southern area of the SPA and large northern area during Survey 1. This equated to 1053 birds (± 95% CI 751 – 1409) and 226 birds (± 95% CI 120 – 347) respectively. During Survey 2, density was calculated at 0.86 birds/km², 0.27 birds/km² and 0.16 birds/km² in the southern part of the SPA, large northern and small northern areas respectively. This equated to 1889 birds (± 95% CI 1485 – 2357) in the southern area of the SPA, 329 birds (± 95% CI 230 – 429) in the large northern area and 61 birds (± 95% CI 27 – 104) in the small northern area.

82 Guillemot *Uria aalge* were recorded in all areas in both surveys. In the southern area of the SPA, density was calculated at 0.02 birds/km² in Survey 1 and increased to 0.95 birds/km² in Survey 2. This equated to 2091 birds (± 95% CI 1180 – 3131) in Survey 2. Across the southern part of the pSPA, the density estimate was the same in Survey 1 as in the SPA (0.02 birds/km²) but somewhat lower (0.90 birds/km²) in Survey 2. In the large northern area, guillemot density was calculated at 0.04 birds/km² in Survey 1 and increased to 0.24 birds/km² in Survey 2. These equated to 51 birds (± 95% CI 19 – 91) and 288 birds (± 95% CI 151 – 438) respectively. In the small northern area, density was calculated at 0.76 birds/km² in Survey 1 and decreased to 0.19 birds/km² in Survey 2. This equated to 294 birds (± 95% CI 149 – 441) in Survey 1 and 72 birds (± 95% CI 41 – 108) in Survey 2.

83 Razorbill *Alca torda* were recorded in all areas apart from the small northern area in Survey 2. Relatively low numbers were recorded across the surveys with the lowest density estimate at 0.02 birds/km² in the large northern area during Survey 2 and the highest density estimate at 0.13 birds/km² in the southern area of the SPA on Survey 2. This equated to 291 birds (± 95% CI 171 – 419) in Survey 2 in the southern area.

84 Grey seal *Halichoerus grypus* were only recorded in the southern area of the SPA/pSPA in both surveys with density calculated at 0.01 animals/km² and 0.02 animals/km² in Survey 1 and Survey 2 respectively. This equated to a population estimate of 14 animals (± 95% CI 0 – 32) and 47 animals (± 95% CI 20 – 77) respectively.

85 Harbour seal *Phoca vitulina* were only recorded in Survey 2 in the southern area in the SPA/pSPA with a low density calculated at 0.02 animals/km² equating to 40 animals (± 95% CI 13– 47).

86 Harbour porpoise *Phocoena phocoena* density estimates were higher in Survey 2 than in Survey 1: in the southern area of the SPA density was calculated at 0.08 animals/km² in Survey 1 and 0.21 animals/km² in Survey 2 which equated to 166 animals (± 95% CI 71 – 279) and 458 animals (± 95% CI 241 – 685) respectively. In the pSPA southern area, the species density estimate was calculated at 0.07 animals/km² in Survey 1 and 0.18 animals/km² in Survey 2 which equated to 164 and 427 animals respectively. Within the large northern area, density was calculated at 0.02 animals/km² in Survey 1 equating to 21 animals (± 95% CI 0 – 60) and 0.14 animals/km² in Survey 2 equating to 175 animals (± 95% CI 80 – 290). In the small northern area, harbour porpoise density was calculated at 0.03 animals/km² in Survey 1 increasing to 0.34 animals/km² in Survey 2. This equated to 11 animals (± 95% CI 0 – 29) and 132 animals (± 95% CI 65 – 217) respectively. A summary table for this species is shown in Table 31.

5.3 Distribution patterns

87 The distribution patterns of the most abundant species and species groups are presented as density maps, derived by applying kernel density estimation, in which a density surface depicts the estimated number of individuals per km² (Figure 7 to Figure 24 and Figure 27). Fuller details of bird numbers are presented in Table 6 to Table 32.

88 Species or species groups for which there were too few observations to generate density maps are presented as dot maps only (Figure 25 to Figure 26 and Figure 28 to Figure 29). These are provided for reference only.

89 Anthropogenic activity is presented as dot maps for reference only (Figure 30 to Figure 32). These include wind turbines, fishing buoys, and boats. This is largely confined to the southern sectors with a coastal bias to most objects. The AIS receiver covered a recording sweep of at least 40km from the aircraft. An AIS device was only operating in the aircraft that flew the southern sector, but this did provide data for much of the northern sector also. Further offshore are three wind farms - the London Array, Kentish Flats and Gunfleet Sands sites.

90 In all of the distribution maps, the Foulness Danger Area 'Hole' can be seen to the east of the Essex coast, at the north-eastern limit to the intertidal flats off Foulness. Although this area was overflown on two transects on each survey (Figure 2), records of birds seen in this area were not included in deriving population abundance estimates or producing density maps because, for reasons explained in the footnote to Figure 1, the area lies outwith the boundary of the newly re-classified Outer Thames Estuary SPA.

91 Red-throated diver (Figure 7 to Figure 10) was the most abundant species during both surveys and showed high densities and a widespread distribution. In the northern sector birds were further from the coast in the first survey than the second, when densities also increased. The distribution was similar in both surveys for the southern zone, but densities showed a strong increase notable in waters either side of the shipping lanes and the London Array wind farm. Such concentrations, where birds are squeezed between areas of human activity, strongly suggests that the birds are undertaking displacement behaviour. While still favouring the area as a whole they are being pushed away from some localities and clustering as a result. This is particularly notable during the second survey.

92 Common scoter (Figure 11) numbers were low with no clear spatial pattern to the records. In the second survey numbers were slightly higher with two small groups recorded near Aldeburgh and another at the far east of the survey area (where they associated with a surf scoter).

93 Gannet (Figure 12) were only recorded in the second survey with records typically away from the coast and with a strong bias to the most offshore part of the southern zone. No gannets were recorded within the footprint of London Array windfarm.

94 Cormorant (Figure 13) show several density hotspots off Dengie National Nature Reserve in Essex and to the north offshore from Walton-on-the Naze. Lower numbers occurred consistently at the mouth of the Thames. There appeared to be no attraction to the London array wind farm.

95 Great crested grebe (Figure 14) favoured the southern Kent coastal area. In the north a scatter of records occurred around Aldeburgh during the second survey.

96 Kittiwake (Figure 15) is a typically pelagic species so the distribution of records in survey one was as expected with low densities. Densities dropped further in the second survey period with birds also then being found further offshore. These movements may be weather related with the first survey following some windy conditions which may have displaced birds into the survey area while in the second period a more typical distribution was maintained.

97 Black-headed gull (Figure 16) showed similar patterns in both survey rounds. Very few birds were seen in the north, with concentrations in the south close to the north east Kent coast and around Southend-on-Sea and off Shoeburyness.

98 Common gull (Figure 17) were present on both surveys across much of the survey zone. There was a minor increase in density between the first and second surveys in the north with the distribution widening slightly also. This increase was mirrored in the south with birds more widely scattered in the second survey (than in the first survey) when their distribution was less coastal than that of black-headed gulls but more coastal than that of kittiwake.

99 Lesser black backed gull (Figure 18) were rare on the first survey, but numbers strongly increased in the second survey with a widespread, scattered distribution of birds at low density. This pattern is likely to be linked with the species being an early spring migrant and birds returning from more southern latitudes to UK waters in mid-February.

100 Herring gull (Figure 19) showed linkage to coastal towns during the first survey. In the second survey this pattern was diluted, and birds were more dispersed across the entire southern sector although still with a bias towards coastal areas. A similar pattern emerged in the northern zone.

101 Great black-backed gull (Figure 20) showed a widely scattered distribution over both surveys and all sectors. Most records related to low numbers of birds with a slight skew towards shipping lanes in the southern sector. Birds seem to also avoid the area around the London array wind farm.

102 Guillemot (Figure 21) were largely absent during the first survey bar a cluster of records in the small north east sector of the northern survey zone. Numbers in this area had dispersed by the second survey but there was a strong bias to offshore waters at the east of the survey site in the southern sector.

103 Razorbill (Figure 22) were recorded with greater frequency during the second survey, especially in eastern sectors of the southern survey zone. Birds were at low density and showed signs of displacement from the three constructed wind farms.

104 Harbour porpoise (Figure 27) were recorded in the more pelagic areas of the site. Animals were always at low densities but clearly showed a bias towards the eastern survey areas during the second survey round. Animals would appear to show displacement effects from the London Array wind farm.

105 Seal species (Figure 28 and Figure 29) records showed a bias towards the southern survey zone, with animals distributed across a wide area. Most of the individuals that received a positive identification (Figure 28) were noted as grey seals.

5.4 Use of the extended part of the SPA by red-throated divers

106 The pSPA includes a section of coastal mudflats off Foulness which are flooded at high tide. Birds were recorded in the outer fringe of this area on both surveys (Figure 33 and Figure 34). No surveys were flown at low tide.

107 The number of red-throated divers estimated to occur in the recently extended part of the SPA around Foulness was calculated by selecting the grid cells from the kernel density estimation plots of their distribution and calculating the population size from the mean density times the area of the selected cells. These more accurate but deterministic estimates of the number of individuals is presented in Table 33.

Table 33 Red-throated diver population estimates for the area of the SPA extension area only, derived using kernel density estimation density calculations.

Survey	SPA Extension only (population estimate)
Survey 1	101
Survey 2	229

5.5 Behaviours of seabirds and non-avian animals

108 The behaviour of seabirds has been categorised as follows: flying and sitting. The number of each observed is presented in Table 34 to Table 37. In addition, the surfacing behaviour for all non-avian animals is presented in Table 38. Snapshot surfacing indicates where the head of a seal or dorsal fin of a cetacean is clear of the water surface in the middle frame of the sequence in which the animal is present. Surfacing is defined as when the animal breaches the surface on a frame which is not in the middle of the sequence where the animal is present.

109 Low numbers of red-throated diver were recorded flying with 13% and 4% recorded flying in Survey 1 and 2 respectively in the southern area and 1% and 3% flying in the combined northern areas. This indicates the flight height of the plane is successful in not flushing this sensitive species.

6 Discussion

110 The surveys were successful in characterising the bird and mammal species present in the Outer Thames survey area, recording a total of 7906 birds of 23 species and 138 marine mammals of 3 species. A further 572 animals were recorded which were not assigned to a named species.

111 The survey extended over the open shore areas of Foulness which at the outset of the project were part of the pSPA but have since been included within the revised boundary of the newly re-classified Outer Thames Estuary SPA. However, some other more enclosed areas out with the original SPA boundary but also now included within the revised SPA boundary, that is the sections within the Rivers Crouch and Roach, and Rivers Yare and Bure and River Blyth, were not surveyed. These sections omitted from our study area for digital aerial surveys can be counted more effectively from land under the WeBS programme. For the sake of completeness, the small 'hole' off Foulness that is now surrounded by the new SPA boundary was flown and the imagery analysed, but the numbers recorded in this area were not included in the abundance estimates.

112 Surveys were originally programmed for the months of January and February when red-throated diver density in the SPA is known to be highest (Webb and others 2009). Restrictions applied through a military flight zone and poor weather in January resulted in the initial survey slipping into early February. Both surveys ultimately took place that month, 13 days apart.

113 At tender stage HiDef noted the presence of aviation danger areas D138 and D138C located around Shoeburyness and Foulness which are often active up to 6000 ft. asl. This was known to be a restriction and dialogue with the relevant parties began early. Despite negotiation it was only possible to attain flight consent at weekends when the danger area was deemed safe. This combined with weather issues subsequently meant that opportunities to survey had to be taken when they arose.

114 The survey design consisted of 42 transects and achieved 13% coverage of the site overall; 15% in the high density southern stratum and 10% in the low-density stratum in the northern sector. This contrasted with the 2013 method of survey which was flown on a series of transects separated by 1.8 km, collecting abutting 3 cm resolution imagery, providing average coverage of 15%.

115 To minimise the effect of movement of red-throated divers between transects and different sectors, HiDef made a commitment to complete all survey in a single day by deploying two aircraft on the same day. This phenomenon of red-throated diver movements in the Outer Thames was highlighted by Webb and others (2009) and could have potentially caused significant bias to any abundance estimates from surveys that did not minimise this effect. The 2013 survey undertook surveys over two dates in January and three dates in February. As such there was potential for movement of birds between dates and a subsequent potential loss of accuracy in the ability to provide a clean snapshot of red-throated diver numbers.

116 The number of red-throated divers recorded in the survey imagery was extrapolated to the SPA boundary, using design-based analysis methods. Density and population estimates, as well as the upper and lower 95% confidence limits were derived by way of a blocked bootstrapping technique.

117 Only one diver species, red-throated, was recorded during the surveys. They were present in high numbers and high densities, particularly in the southern sector of the survey zone.

118 Survey results are presented in Table 6 to Table 29. A summary table is provided in Table 4. The peak estimate of red-throated diver within the original SPA boundary occurred in the second survey on 17 February 2018; that is 21,997 individuals. Considering also the additional area off Foulness that is now included within the revised boundary of the newly re-classified SPA, the peak estimate of red-throated divers across the entire extended SPA also occurred in the second survey; that is 22,280 individuals These figures represent more than 340% of the population for which the Outer Thames Estuary SPA was first classified in 2010; that is 6,466 individuals.

119 We estimated the population size of red-throated divers in the small area of sea off Foulness that is now included within the recent extension to the OTE SPA using a deterministic method based on the numbers estimated using kernel density estimation. While this method gave a more sensible population estimate than the stochastic transect-based method, it gave no confidence limits to this sub-sample and couldn't be used to adjust the population estimates of red-throated divers within the original SPA boundary. A better approach would have been to use a stochastic spatially-explicit modelling method to estimate the population size, such as the MRSea suite of modelling tools (Mackenzie and others 2013).

120 Over the years various attempts have been made to estimate the red-throated diver population in the Outer Thames. The results of these are summarised in Table 39 and show the overall trend for increased populations.

Table 39 Summary of known recognised estimates of red-throated diver populations in the Outer Thames (accounts derived from Webb and others 2009 and APEM 2013))

Survey date	Red-throated diver population estimate	Lower 95% confidence limit of population	Upper 95% confidence limit of population	Survey method used	Comment on surveys
January 2002	2,460	1,667	3,630	Visual aerial survey	Covered Greater Thames area >900,000ha.
January 2003	10,884	8,115	13,439	Visual aerial survey	Covered Greater Thames area >900,000ha.
February 2004	7,688	5,041	11,725	Visual aerial survey	Covered Greater Thames area >900,000ha.

Survey date	Red-throated diver population estimate	Lower 95% confidence limit of population	Upper 95% confidence limit of population	Survey method used	Comment on surveys
Jan/Feb 2005	6,123	4,996	7,504	Visual aerial survey	Covered Greater Thames area >900,000ha.
January 2006	5,291	3,745	7,179	Visual aerial survey	Covered Greater Thames area >900,000ha.
Feb/March 2007	3,106	2,035	4,589	Visual aerial survey	Covered Greater Thames area >900,000ha.
Peak mean over the period 1989-2006/07	6,466			Visual aerial survey	Outer Thames SPA (a smaller subset of the Greater Thames) c.379,268ha. Calculated by averaging peaks in annual estimates in numbers. Webb and others (2009).
January 2013	11,248	8,649	14,155	Digital still aerial survey undertaken over two days (3cm resolution)	Outer Thames SPA (a smaller subset of the Greater Thames) c.379,268ha. APEM (2013)
February 2013	14,161	8,230	22,245	Digital still aerial survey undertaken over two days (3cm resolution)	Outer Thames SPA (a smaller subset of the Greater Thames) c.379,268ha. APEM (2013)

Survey date	Red-throated diver population estimate	Lower 95% confidence limit of population	Upper 95% confidence limit of population	Survey method used	Comment on surveys
February 2018 (survey 1)	10,136	7,763	12,626	Digital video aerial survey undertaken on a single day (2cm resolution)	Outer Thames SPA (a smaller subset of the Greater Thames c.388,768ha)
February 2018 (survey 1)	10,148	7,868	12,544	Digital video aerial survey undertaken on a single day (2cm resolution)	Outer Thames pSPA (a smaller subset of the Greater Thames but c95km² larger than the SPA)
February 2018 (survey 2)	21,997	15,351	29,415	Digital video aerial survey undertaken on a single day (2cm resolution)	Outer Thames SPA (a smaller subset of the Greater Thames c.388,768ha)
February 2018 (survey 2)	22,280	15,611	29,784	Digital video aerial survey undertaken on a single day (2cm resolution)	Outer Thames pSPA (a smaller subset of the Greater Thames but c95km² larger than the SPA)

121 These population estimates (Table 39) should still be regarded as minima for all of these surveys given that all of them will have been affected by not all birds being above the sea surface and so available for detection (known as availability bias). In common with all previous studies detailed in Table 39, we made no correction for this affect for red-throated diver or any other diving species, including harbour porpoise. While it is possible to correct approximately for availability bias for many species using known diving rates, we did not do this for red-throated divers because accurate dive rate data are not known to exist.

122 The survey results in Table 39 are not directly comparable because they follow different methods and different sizes of study area. However, they do show large variation in results within similar parameters (that is total population and area) and survey areas, tending

toward an increase in recent years. A full exploration of the variation and apparent increase in recent years would require returning to review original data but the following points, as a minimum, seem likely to be relevant:

- Genuine increase in divers in the survey area over time;
- Changes in anthropogenic activity in the area, notably the introduction of the London Array, Kentish Flats, Gunfleet Sands and Scroby Sands wind farms;
- Changes in environmental variables;
- Changes (improvements) in methodology, that is differences in survey design and analytical techniques;
- Changes in technology utilised in surveys resulting in improved detection rates or reduced levels of bias; that is from boat to aircraft and improved imaging methods.

123 Other species were recorded, and data processed, with results presented, but the focus of the contract relates to red-throated diver. From data collected relating to divers distance modelling was applied and an estimated population for red-throated diver provided.

124 It should be noted that the area of the pSPA (now fully re-classified as a SPA) represents a slight increase over that of the original size of the SPA (as classified in 2010). This increase includes the tidal flats around parts of the Essex coast, of about 9500 ha, which have been included in the SPA primarily in recognition of their importance as foraging grounds for breeding common terns *Sterna hirundo* originating from the colony within Foulness SPA. The surveys did not record divers in this area of exposed mudflats and thereby often unsuitable habitat for divers. However, there were a number of raw observations of red-throated divers in the extension to the SPA in areas where the sea covered the mudflats, and extrapolated data from KDE for this area suggest red-throated diver population estimates of 101 individuals in Survey 1 and 229 individuals in Survey 2 in this area into which the Outer Thames Estuary SPA now extends (Table 33).

125 Counts derived using the same transect sampling regime to calculate the population estimate of red-throated divers in the original (smaller) SPA and the new, extended SPA boundary resulted in a larger point estimate of the population size in the smaller SPA boundary than the larger, encompassing boundary.

126 The first survey took place in weather conditions in which the sea state was greater than that encountered in the second survey flights. While such weather events can impact on digital aerial survey results, it was expected that the later survey would encounter a higher number of birds (Webb and others 2009) given a review of previous survey data and known movements of the species. However, the increase in bird density between the first and second surveys affected most species, including gannets, which are very easily detected regardless of sea state, while a more cryptic species, the kittiwake, was more abundant in the first survey when the sea state was rougher. Sea state in Survey 1 was still within the normal boundaries for surveying.

127 As predicted at the tender stage, the surveys confirmed that the northern section could be deemed to be a low-density survey stratum within a stratified sampling design, and the southern section of the SPA a high-density stratum.

128 The only previous digital aerial surveys of the entire Outer Thames Estuary SPA were conducted in early 2013 by APEM Ltd. (APEM 2013). These surveys yielded design-based estimates of the population of red-throated divers within the original SPA boundary of 11,248 (± 95% CI 8,649 – 14,155) in January 2013 and 14,161 (± 95% CI 8,230 – 22,245) in February 2013. The former population estimate, and its confidence limits, are very similar to those recorded in survey 1 in 2018 (10,136 ± 95% CI 7,763 –12,626). The figure of 14,161 recorded in February 2013 exceeds that recorded in survey 1 in 2018 (10,136) and the upper 95% CI of that (12,626). However, 14,161 is considerably less than the population estimate of 21,997 recorded in survey 2 in 2018 and is indeed less than the lower 95% CI of that population estimate (15,351). The upper 95% CI around the February 2013 population estimate (22,245) is however very similar to the mean population estimate in survey 2 in 2018 (21,997). Thus, broadly speaking the results obtained in 2013 and 2018 are reasonably consistent.

129 In combination with the estimated red-throated diver populations within the newly enlarged SPA of 10,148 (Survey 1) and 22,280 (Survey 2), the population estimates for all other species/ species groups combined yield total estimated abundances across this area of 19,071 individuals (Survey 1) and 46,056 individuals (Survey 2). For the wintering waterbird assemblage within a SPA to be considered as a qualifying feature of that site, UK SPA selection guidelines require that the peak count must exceed 20,000 individuals on a regular basis (JNCC 1999). The results of the 2018 surveys indicate that at least on one occasion the SPA supported an assemblage of wintering waterbirds well in excess of the size required to consider this assemblage to be a qualifying feature of the SPA.

130 Population estimates were derived from mean bootstrapped density which gives mostly small variations in values. This can lead to small anomalies in which the sum of populations in the sub-sections of the SPA do not add up to the estimate obtained for the whole SPA, and in which the smaller site boundary for the original SPA has a larger population estimate than for the larger extended SPA. In these cases, the population estimate from analysis for the entire area, and not the sum of parts, should be used in further management of the site.

6.1 Changes in survey technique and methods – an evolving process

131 In 2018, the survey transects were flown perpendicular to the coast. Effectively this was north to south across the southern sector, and south-east to north-west across the northern zones. This was in contrast to the 2013 survey pattern which was flown south-west to north-east across the southern sector and north-south across the northern zone. While this, and the fact that both surveys were flown on a single day, might be reasons for an increase in the peak population estimates for red-throated divers in the SPA between 2013 and 2018, the peak population estimate for 2018 (21,997) is within the confidence limits of the 2013 peak survey (8,230 – 22,245).

132 HiDef recorded no other diver species apart from red-throated but other diver species were reported consistently during surveys in 2013, with 222 black-throated diver and 248 great northern diver estimated in a single survey (APEM 2013). This result, if proved to be a regularly occurring concentration of these species would justify inclusion of these species in the SPA citation. Webb and others (2009) also found that the presence of these species in visual aerial surveys and in shore-based surveys was rare. It is highly likely that this difference was due to improved identification techniques utilised in 2018 surveys and suggests that reports of the occurrence in 2013 of these species, in such numbers, was erroneous.

133 The technique deployed by HiDef differs from that utilised in 2013 in that HiDef uses digital video rather than digital stills technology. In terms of identification this allows 5 – 7 still images of an object to be reviewed from slightly different angles as the aircraft passes. This contrasts with a single, plan view achieved via the digital stills process. Overall, video provides the identifying team with greater opportunity, and thereby increasing certainty, to come to a conclusive identification.

134 Video footage was attained as 2cm GSD, rather than 3cm GSD as in surveys five years previously. Ground Sample Distance (GSD) is the area of ground that is shown per pixel. The lower the GSD, then the better resolution, but there is a trade-off in other areas of photographic quality once GSD moves below 2cm. This step change in resolution, from 3 to 2, allows identifiers to make more accurate identifications and reduces risk related to errors in identification.

135 Prior to the introduction of digital aerial surveys, visual aerial surveys were flown. These comprised of observers viewing from the aircraft as it flew over the survey zone. This method was introduced to overcome the known bias resulting from red-throated divers avoiding boats (for example Schwemmer and others 2011; Camphuysen and others 2004) and so leading to under recording. A drawback of both these methods is the lack of ability to review or audit any data collected: "*what's hit's history, what's missed is mystery*".

136 Overall, digital video aerial survey provides a more robust data source, as the aircraft flies at a height which negates disturbance issues; the low level of divers noted in flight in 2018 surveys suggest these were not flushed due to the aircraft flying at a higher altitude, while providing a higher degree of confidence in identifications, which can be audited at a later date if queries arise. This degree of confidence is higher as video produces multiple images of the same object compared to the single image of digital still.

6.2 Changes in distribution

137 Shipping data were collated at the time of the survey via an AIS device in one aircraft. This allowed shipping movements on the day of survey to be followed (Figure 30). Additional anthropogenic activity was plotted (Figure 31) and compared visually against species distribution patterns. This suggested that the distribution of a number of species appeared to be affected by human activity in the area.

138 Red-throated divers clearly show displacement from wind turbines and shipping lanes (Figure 8 to Figure 10) but quantifying these movements is out with the scope of this contract.

139 Dierschke and others (2017) raised the issue of displacement and energetics and the consequences which may follow for individuals and also at a population level. Given the construction and operation of several offshore windfarms within the Outer Thames Estuary SPA in recent years and the evidence of marked displacement of divers within this study (and as reported in many previous studies) on the one hand and the simultaneous rise in population estimates of red-throated divers across the SPA as a whole on the other hand, further work is needed on this issue.

140 Displacement has been shown to be a factor influencing the distribution of red-throated divers at other sites with wind farm infrastructure in place (Webb and others 2017; Petersen and others 2014; Dierschke and others 2017). Given the increase in wind farms in the southern North Sea, both within and out with UK waters, it may be that birds are being

displaced over a wider area than previously thought and larger, more focussed aggregations are now forming in the remaining areas of suitable habitat.

141 Figure 35 and Figure 36 show preliminary analysis relating to mean density and abundance of divers from the two surveys in relation to distance from shipping lanes. These show some consistency in aggregation of birds in areas away from shipping traffic (and possibly other man-made structures). Highest densities of birds occur c.5-8km from shipping lanes. This initial output warrants further interpretation and modelling. It is interesting to note that the pattern found here of the density of divers increasing over the first few kilometres from shipping lanes to reach a peak at some distance, followed by a decline from those peak densities in areas even further from shipping lanes was also reported in an analysis of similar survey and shipping data in Liverpool Bay SPA (Burt and others 2017) albeit that in that study peak densities occurred at a distance of c 2km.

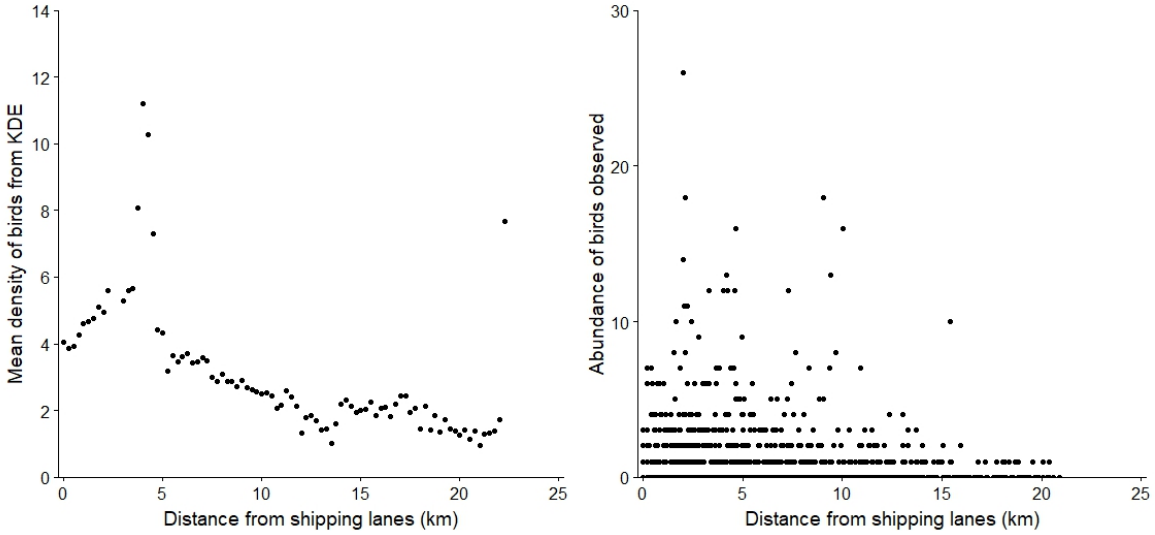


Figure 35 Red-throated diver distance to shipping survey 1. The left graph shows the mean density per grid cell from the KDE surface modelling and on the right is the absolute numbers for each and every transect segment.

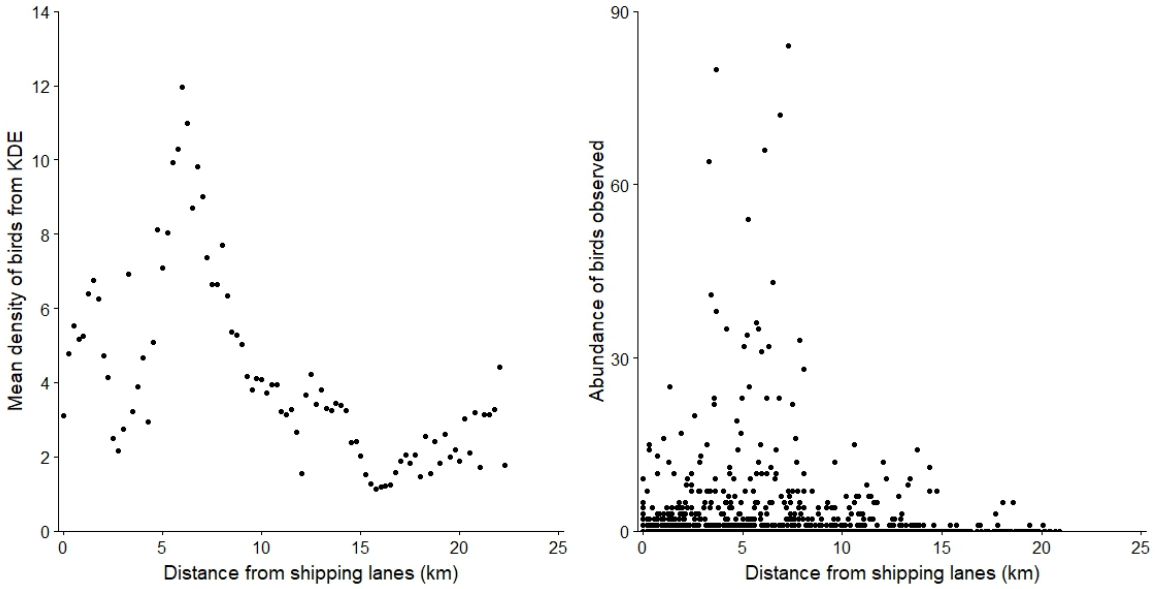


Figure 36 Red-throated diver distance to shipping survey 2. The left graph shows the mean density per grid cell from the KDE surface modelling and on the right is the absolute numbers for each and every transect segment.

6.3 Recent survey work elsewhere in the southern North Sea

142 Recent work in the southern North Sea (http://bioconsult-sh.de/en/projects/loon-radio-telemtry/) has seen divers caught and equipped with satellite transmitters on their wintering area within the German Exclusive Economic Zone (EEZ). A considerable volume of data has been collected and is currently being fully analysed (reporting due in Q3 2018). Considering offshore windfarm developments, spatial distribution and temporal characteristics of habitat use of divers will be analysed to set a basis for the relation of habitat loss due to offshore windfarms and habitat requirements of divers. Outcomes form this project seem likely to assist in discussion of displacement of birds in the Outer Thames area and elsewhere in UK waters.

7 Conclusions

143 The surveys were successful in characterising the bird and mammal species present in the Outer Thames survey area, recording a total of 7906 birds of 23 species and 138 marine mammals of 3 species. A further 572 animals were recorded which were not assigned to a species, an identification rate to species level of 92.53% across the survey programme.

144 In line with the core contract objectives two new population estimates of 10,136 (Survey 1) and 21,997 (Survey 2) red-throated divers are provided within the original Outer Thames Estuary SPA boundary, or of 10,148 (Survey 1) and 22,280 (Survey 2) within the somewhat larger boundary of the newly reclassified SPA.

145 Updated population estimates of other species are provided (Table 32), some of which could be considered in any future review of the qualifying features of this SPA. The large estimated population of red-throated divers combined with the population estimates for all the other species or species groups yields a peak abundance across all bird species within the boundaries of the newly re-classified SPA of 46,056 individuals. This is more than double the threshold size set out in the UK SPA selection guidelines for an assemblage to be considered as a qualifying feature of a SPA.

146 Consideration should be given as to whether other marine SPAs recently designated for divers, grebes, scoters and other sea-ducks (for example Greater Wash SPA), or pSPAs that are yet to be classified (for example Solway Firth pSPA) would benefit from new, post-classification surveys using modern survey methods, not least to ensure that their baseline populations are re-estimated using the most up-to-date approaches and technology. Such surveys would also allow the updating of national, and international, population numbers and provide a sound basis for the setting of Conservation Objectives for these newly classified sites.

147 This study provides a basis for additional analysis to determine the effects of different anthropogenic activities in the SPA, such as shipping and the locations of wind farms on the distribution of key species of interest, notably red-throated divers.

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Short-Term Behavioural Responses of Wintering Waterbirds to Marine Activity

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Quantifying the Sensitivity of Waterbird Species during the Non-Breeding Season to Marine Activities in Orkney and the Western Isles

SHORT-TERM BEHAVIOURAL RESPONSES OF WINTERING WATERBIRDS TO MARINE ACTIVITY

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Report of work carried out by the British Trust for Ornithology on behalf of the Marine Scotland

April 2018

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Short-Term Behavioural Responses of Wintering Waterbirds to Marine Activity (CR/2015/17)

D Jarrett, A S C P Cook, I Woodward, K Ross, C Horswill, D Dadam and E M Humphreys

British Trust for Ornithology

1. Executive Summary

1. Waterbirds are subject to a range of pressures on their wintering grounds, with disturbance and pollution being potentially the most significant. In the British Isles, significant populations of waterbirds winter in the seas including around Orkney and the Western Isles.

2. Increased marine activity associated with exploitation of the abundant wind, wave and tidal resources around Orkney and the Western Isles, in addition to increased aquaculture activity and existing shipping and shellfishing activity, has the potential to negatively impact wintering waterbird populations through increased disturbance.

3. The scope of this research project was to compare the relative sensitivities to marine activity of eleven target waterbird species during the non-breeding season. This was achieved by gathering data during a single winter fieldwork season on Orkney.

4. The research was focused on the following species: Common Eider, Long-tailed Duck, Velvet Scoter, Common Goldeneye, Red-breasted Merganser, Red-throated Diver, Black-throated Diver, Great Northern Diver, European Shag, Slavonian Grebe and Black Guillemot.

5. Data were collected using three complementary survey methodologies: Vantage Point (VPs) surveys, focal flock watches and ferry surveys. The methods were designed to gather systematic data on the target species' short term behavioural responses to marine activity, and environmental variables considered likely to affect those responses.

6. The VP surveys indicated that Common Eider, Long-tailed Duck and European Shag were all significantly more likely to fly in the presence of

marine activity. The numbers of Common Eider, Long-tailed Duck, European Shag and Great Northern Diver within a pre-defined study area all declined following marine activity, whereas Black Guillemot numbers appeared unaffected.

7. The focal flock watch methodology involved recording the behaviour of target species flocks in the presence and absence of marine activity. Long-tailed Duck flock size was likely to decrease in the five minute period following a disturbance event, and this species was also the most frequently recorded flying in the absence of marine activity. Great Northern Diver and Black-throated Diver were very unlikely to fly either in the presence or absence of marine activity.

8. The ferry survey methodology involved gathering data on regular island ferry services and recorded target species' responses to the passing ferry. Red-throated Diver, Black-throated Diver and Slavonian Grebe were the most likely species to exhibit a response (flight, evasive swim, or dive) to the passing ferry, and Red-throated Diver was most likely to show a flight response.

9. Whilst data were successfully gathered for nine of the target species across the three survey methodologies, too few data were collected for Common Goldeneye and Velvet Scoter to be able to draw conclusions.

10. Combining data gathered across the three methodologies, we categorised the sensitivity of nine target species (excluding Velvet Scoter and Common Goldeneye) as follows: **very high** (Red-throated Diver, Black-throated Diver, Slavonian Grebe and Red-breasted Merganser); **high** (Long tailed Duck and Great Northern Diver); **medium** (Common Eider and European Shag) and **low** (Black Guillemot). These sensitivities are assessed only in relation to the other target species, and are based solely on the data on short-term responses to marine activity gathered during this project.

11. Prior to this research little information on sensitivity to disturbance was in the public domain for Great Northern Diver, Slavonian Grebe and Black-throated Diver. The results presented here address important knowledge gaps which will help inform the marine planning process.

12. Gathering robust data on the effect of marine disturbance on waterbirds is challenging. Whichever methodological approach is followed there are likely

to be significant practical challenges in data collection and analyses, and limitations on the conclusions which can be drawn. Furthermore, the spatial and temporal scales at which data is collected and analysed can significantly influence the results obtained.

13. Although all the fieldwork took place in Orkney, it is likely that the findings presented here are also applicable to the Western Isles. However, we would advise against conclusions made here being applied to the larger estuarine ports of mainland Britain, where marine activity is likely to be of a different scale and intensity.

14. It should be borne in mind that the results and assessments presented in this report are based on short-term behavioural responses. Research is needed to better understand the mechanisms by which short-term behavioural responses might translate into demographic effects and the relative significance of the disturbance effects on habitat loss and energetic expenditure should be considered.

15. Recommendations for future research include suggestions to carry out a before-after-gradient study on the effect of new marine developments to assess long-term displacement effects, tracking studies of waterbird use of the marine environment in relation to marine activity, and trial disturbance tests using chartered boats.

7.4. Methodological Insights and Recommendations for Further Work

Gathering robust data on the effect of marine disturbance on waterbirds is challenging. Whichever methodological approach is followed there are likely to be significant practical challenges in data gathering, and limitations on the conclusions which can be drawn. As such it is critical that methodologies should be trialled and refined in field conditions to ensure they are feasible prior to data collection commencing. We present the following insights gathered during this project as a potentially valuable source of information for those planning similar work.

The results from the ferry surveys and the results from Schwemmer et al. (2011) provide evidence that in rougher sea conditions disturbance effects are likely to be greater. However, when using shore-bound survey methods it becomes challenging to gain accurate counts in sea conditions from a sea state three upwards (assuming count areas extend to 1 km or greater). This means that during periods when birds may be most vulnerable to disturbance shore-bound surveys will either not be possible or the accuracy of the data gathered will be compromised.

Shore-bound surveys must be focused on areas of sea which are subject to regular marine activity, because if this is not the case, then little or no data on disturbance responses can be gathered (speculatively waiting for marine activity in areas of sea where activity rarely takes place is not a fruitful approach). However, it is probable that species (and individuals within species) that are most sensitive to disturbance will avoid areas of sea which are frequently disturbed by marine activity. Therefore, conclusions made on species' sensitivities based on data gathered in areas of sea subject to regular marine activity may not be transferable to relatively undisturbed areas. Experimental tests, discussed in Section 7.4.2, may be the only solution to this problem.

From the VP surveys we were able to detect a statistically significant effect of marine activity on target species for those species which were most abundant in regularly disturbed areas and therefore had the largest datasets available. However, the fact that these species were abundant in frequently disturbed areas may indicate that they were in fact less sensitive to disturbance. Therefore, it may be the case that those species for which we were able to detect significant effects on flight activity or changing abundance following disturbance using the VP survey methodology were actually *less sensitive* to disturbance than species for which we couldn't detect an effect.

Ferries follow regular routes so data gathering also took place in areas with regular marine activity, ferry surveys do have the benefit that relatively large datasets can be generated relatively quickly and cost-effectively (Larsen and Laubek 2005). It is also possible to gather data up to a sea state five because it is only necessary to detect birds up to 300 m from the boat (and only a sample is required). Schwemmer et al. (2011) found that the FID (flight initiation distance – the distance from the approaching boat at which birds took flight) varies significantly by species, and recorded instances of Common Scoters flushing at distances of up to 3.3 km. In this research, though we didn't systematically record FID, we didn't observe birds flushing at greater than approximately 1 km. While we are confident that few flight responses were missed, it is harder to judge whether swim and dive responses in advance of the ferry may have been missed. Great Northern Diver swim responses were regularly recorded and recorded up to 2 km ahead of the ferry, and on very calm days were recorded swimming out the path of the ferry at greater distances (up to approximately 4 km on occasion). However, the possibility that some distant responses were unrecorded is unlikely to affect the overall pattern of the data or lead to incorrect conclusions being drawn.

7.4.1. Suggested Alterations to Methodologies used During this Research

VP Surveys

The analysis of the VP survey data demonstrates that if the spatial scale of the analysis is inappropriate, then it may not be possible to accurately detect disturbance effects. We gathered data in sectors of four different sizes, and in the largest and smallest sectors we did not generate meaningful data on disturbance effects, as shown in Table 23. The size of the L1/R1 sectors was likely to be too small to detect disturbance effects, and too close to shore (where few disturbance effects took place at most VPs). The size of the L4/R4 sectors were likely to be too large to detect disturbance effects, with the size of the sector meaning that birds within the sector were less likely to interact with vessels, and when they were disturbed were less likely to leave the sector than birds within smaller sectors. There would also have been reduced count accuracy in the L4/R4 sectors due to the distance from the fieldworker. The only species for which including all sectors appeared to improve the modelling of change in abundance was Great Northern Diver. This will be because of the higher detectability of this species at distance, and because almost 80% of Great Northern Diver records were in the L4/R4 sectors, so including these sectors in the analysis increases the size of the dataset significantly.

There are clear trade-offs involved in data gathering methods: surveying a larger study area increases the amount of interactions between vessels and birds recorded, but requires more time be spent counting birds, which reduces the frequency with which birds can be counted. If the frequency with which counts are made does not correspond to the temporal pattern of responses to marine activity, then the impact of disturbance on birds may be missed or underestimated. We would recommend that recording should be limited to a distance of 1 km or less from the fieldworker to reduce counting time, and, therefore, enable increased frequency of counts – the VP surveys show that for most species, detecting an effect of marine activity on abundance of birds is easier when shorter time periods are analysed.

The behaviour, abundance and distribution of target species should be considered when designing VP surveys. For example, Velvet Scoters and Black-throated Divers are rarely recorded within 1 km of shore (per field obs), so shore-bound VP surveys may not be effective for these species.

Table 23: Comparisons of sectors used in VP survey analysis. Area relates to one individual sector (e.g. L4, rather than L4 and R4 combined). *Based on comparing the results of models which included only L2, R2, L3, R3, and models which included all sectors for Eider, Shag, Long-tailed Duck.

Sector	Distance from fieldworker (m)	Area of sector (m^2)	Relative strength of disturbance as effects detected
L4 / R4	1,000 - 2,000m	1,178,097	Weak*
L3 / R3	500 - 1,000m	294,524	Strong*
L2 / R2	250 - 500m	73,631	Strong*
L1 / R1	0 - 250m	24,544	Weak*

Focal Flock Watch Surveys

The propensity of some species to fly in response to marine activity can drop by as much as 50% from the 0-50 m to the 50-100 m distance band (Figure 21). However, when carrying out focal flock watches, the proximity of birds to the vessel could often not be accurately estimated. As such, the most significant variable determining response (distance to marine vessel) was not effectively recorded during flock watch surveys. This limitation will have reduced the effectiveness of the focal flock watch surveys in analysing behavioural responses.

The eleven different target species of this study have different flocking, feeding and evasive behaviours. Applying one methodology to assess disturbance impacts across all target species which in theory should allow comparison between the target species, may in fact limit the degree to which responses to disturbance can be recorded if behavioural responses vary significantly between species. The time periods within which behaviour was recorded (five minutes in this research) should also ideally be adapted for different species - there may be some species for which the behavioural consequences of disturbance occur over a longer or shorter time period, and thus are not picked up with five-minute recording periods. And indeed, the results from the VP surveys do suggest that the temporal scale of responses varies across the target species. Targeted survey methods informed by knowledge of individual species behavioural characteristics may be more effective in gathering meaningful data on behavioural responses to marine activity.

7.4.2. Recommendations for Future Research

The following recommendations for future research on waterbird sensitivity to marine activity are made:

A) Experimental Disturbance using Chartered Boats

The trial disturbances of target species using a chartered boat using a standardised approach protocol would be a time-efficient means of gathering data in areas not subject to regular marine activity. Depending on resources, it would also be possible to test the effect of the frequency and pattern of disturbance events. The significant advantage of chartering a vessel for experimental tests is that researchers would be able to select the areas for the experiments to be carried out, and would be able to make effective comparisons between responses in areas of sea subject to regular marine activity and areas of sea where marine activity is absent. This approach would also allow the gathering of systematic data on scarcer species which may avoid areas where regular marine activity is present.

B) Displacement Studies

An important research need is to gather evidence on the extent to which increased levels of marine activity can lead to long-term displacement effects. A multi-year study using a before-after-gradient (BAG) design (Mackenzie et al. 2013) in an area subject to significantly increased marine activity would address this need. It would also be possible, having carried out a long-term displacement study along these lines, to then refer back to this research and other research on short-term

behavioural responses to examine the extent to which short-term behavioural responses are a reliable means of predicting sensitivity to displacement effects.

C) Tracking Studies

For future studies on the effects of marine activity on waterbirds, using tracking devices to monitor bird movements and activity during the wintering period may be an effective means of gathering data on interactions with marine activity. At a coarse scale, data on marine traffic are available (sometimes at a cost, depending on the type of data required) which could be analysed with data from tracking devices and/or accelerometers to infer behaviour, energetic costs and movements in relation to marine activity. A key priority for work on sensitivity to marine activity should be to gather data on the relative significance of habitat loss compared to the energetic costs of displacement; tracking studies may be the best means to assess this.

Spatial data on smaller vessels is often not available, which would limit the usefulness of this approach. In addition, issues related to long term attachment methods may affect the extent to which tracking studies are currently feasible.

D) Timing of Evasive Flights

An additional piece of data which wasn't a focus of this study but which may be a relatively simple means of gathering comparable data on the sensitivities of target species is the time/distance of evasive flights. This would provide some indication of i) the relative energetic cost of the disturbance event on the target species and ii) the distance the bird has to travel to find an equivalent area of habitat (see Section 7.5, where the importance of this is discussed). Whilst carrying out the focal flock watch surveys fieldworkers were able to gather a small dataset on the timing of evasive flights. We found that Slavonian Grebe evasive flights averaged 47 seconds (five flights), Red-breasted Merganser 28 seconds (6), Long-tailed Duck 20 seconds (11), Eider 18 seconds (7), and Red-throated Diver 12 seconds (3). These data correlate reasonably well with the sensitivities estimated during this project, except for Red-throated Divers, for which a higher result might have been expected (however there were only three timed evasive flights). Although the sample sizes are small, the variations found here between the target species may warrant further investigation.

E) Suggested Priority Species for Further Research

We recommend that Slavonian Grebe, Black-throated Diver and Velvet Scoter be priorities for further research. We assessed Slavonian Grebe and Black-throated Diver as highly sensitive species, however, we noted that these categorisations were based on smaller sample sizes. More data on these species would be useful to confirm these assessments. For Velvet Scoter, there was not enough data to draw any conclusions, however, there was some indication (from a very small dataset) that this species may be likely to take flight in response to marine activity.

7.5. Conclusions

This research has contributed to the body of available information on the target species. We have characterized the short term behavioural responses of wintering waterbirds to marine activity as follows: **very high** - Red-throated Diver, Black-throated Diver, Slavonian Grebe and Red-breasted Merganser; **high** - Long tailed Duck and Great Northern Diver; **medium** - Common Eider and European Shag; and **low** - Black Guillemot.

Our results are broadly comparable to previous assessments of sensitivity to disturbance and displacement effects (Furness et al. 2012; Furness et al. 2013). Where assessments of sensitivities differ, this may be because in previous assessments an absence of field data necessitated a reliance on expert opinion or was based on data gathered on similar species. Previous studies have also tended to focus more on flight responses, while here we considered all evasive responses. It's also worth noting that Red-breasted Merganser has not been included in previous reviews (Furness et al. 2012; Furness et al. 2013; Wade et al 2016); we would recommend they should be considered a relevant species when considering disturbance effects of marine activity on wintering waterbirds. For Great Northern Diver, Black-throated Diver and Slavonian Grebe, little information on sensitivity to disturbance was in the public domain prior to this work (WP1, Appendix 2), so the results of this research will fill important knowledge gaps for these species. Although all the fieldwork took place in Orkney, it is likely that the findings presented here are broadly applicable to the Western Isles in terms of the type and intensity of marine activity and coastal topography, (see WP2, Appendix 3). Because almost all the vessels which were recorded during this study were relatively small (<50 m length), we would advise against the conclusions made here being applied to the larger estuarine ports of mainland Britain for example, where the marine activity is likely to be characterized by much larger vessels.

For those very scarce target species (Common Goldeneye and Velvet Scoter) it was apparent that neither on-shore based surveys (which are reliant on marine disturbance events occurring) nor ferry surveys were an efficient method of gathering data. Trial disturbance of these species using a chartered boat may be the only way to gather data for these species. For Slavonian Grebe and Black-throated Diver for which our datasets were small, there are small but significant wintering populations around the Orkney Islands and the low number of records on the VP and ferry surveys may indicate that these species avoid busier areas.

Waterbirds are likely to be negatively affected by disturbance caused by frequent marine activity through elevated energetic costs. Flight is thought in most cases to be the most costly mode of locomotion among birds (Ellis 1984, Nudds and Bryant 2000), and will vary by species as a function of the relationship between body mass and wing area (Pennycuick 2008). The energetic costs of diving and swimming will also vary between species but measuring this is more challenging (Žydelis and Richman 2015). Gaining a better understanding of the relative energetic costs and the overall time spent carrying out evasive behaviour in response to marine activity will be an important next step towards an appreciation of the potential demographic impact of disturbance. Another long term consequence of disturbance arising from marine activity is displacement from key habitats.

For those species likely to fly in response to disturbance (Red-throated Diver, Long-tailed Duck) energetic costs may be a more significant consequence of disturbance (see Dierschke et al. 2017 for a discussion on Red-throated Diver disturbance). Those species for which the relative energetic costs of flight are high (Great Northern Diver, Black-throated Diver, Common Eider) may be relatively more sensitive to fast moving vessels, where flight may become the only means possible of avoiding collision. However, relying on flight response as a predictor of the importance of disturbance effects is also problematic: for example, Long-tailed Duck and Red-throated Diver are relatively frequent fliers in the absence of marine activity, so sensitivity may be over-estimated if it is measured solely by the likelihood of flight responses to marine activity. We also demonstrated here that Great Northern Divers, which very rarely fly in response to disturbance, do nevertheless experience short-term movements when subject to marine activity (Section 6.1.2, 6.1.3). In such instances where flight response was unlikely but the species did appear sensitive to marine activity (Great Northern Diver, Black-throated Diver), habitat loss may be a more significant mechanism.

While to date there has been limited work on the long-term effects of disturbance effects on seaducks, grebes and divers on wintering grounds, past work on waders

has suggested that frequent disturbance can contribute to reduced survival (Burton et al. 2002, 2006; Burton and Armitage 2005, Davidson and Rothwell 1993). This may relate to time spent avoiding disturbance (Urfi et al. 1996, Stillman & Goss-Custard 2002) or to displacement from preferred foraging areas making it harder to meet their daily energy requirements (Burton et al. 2006). We would also advise caution when interpreting short term behavioural responses of wintering waterbirds to marine activity; birds which appear not to be responding to disturbance may be remaining at a site regardless of negative impacts of disturbance, due to a lack of suitable alternative habitat nearby (Gill et al 2001). This suggests that in order to better understand the consequences of disturbance, we also need to understand the relative importance and availability of areas of habitat within an individual's wintering range.

8. Acknowledgements

We are grateful to Andrew Upton, Jim Williams and Stuart Williams for carrying out fieldwork in often challenging conditions, providing valuable advice on wintering waterbirds around Orkney, and for advising on the methodologies used during this project. We are grateful to Orkney Ferries for allowing us access to the bridge area on ferries to carry out surveys and to Mark Wilson for providing valuable advice on statistical methods. This work was carried out on behalf of Marine Scotland and overseen by a steering group including Lucy Law, Jared Wilson (Marine Scotland), Helen Rowell, Tim Frayling (Natural England), Emma Phillip, George Lees (Scottish Natural Heritage), Matty Murphy (Natural Resources Wales), Alex Sansom (Royal Society for the Protection of Birds), Ross McGregor (Hi-def) and Peter Coldwell (Comhairle nan Eilean Siar). We also thank Alex Robbins (Scottish Natural Heritage) for useful discussions over the course of this project. The work was funded by the Scottish Government's Contract Research Fund.

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For more information about the UK Biodiversity Action Plan visit http://www.jncc.gov.uk/page-5155









BIODIVERSITY

THE UK ACTION PLAN

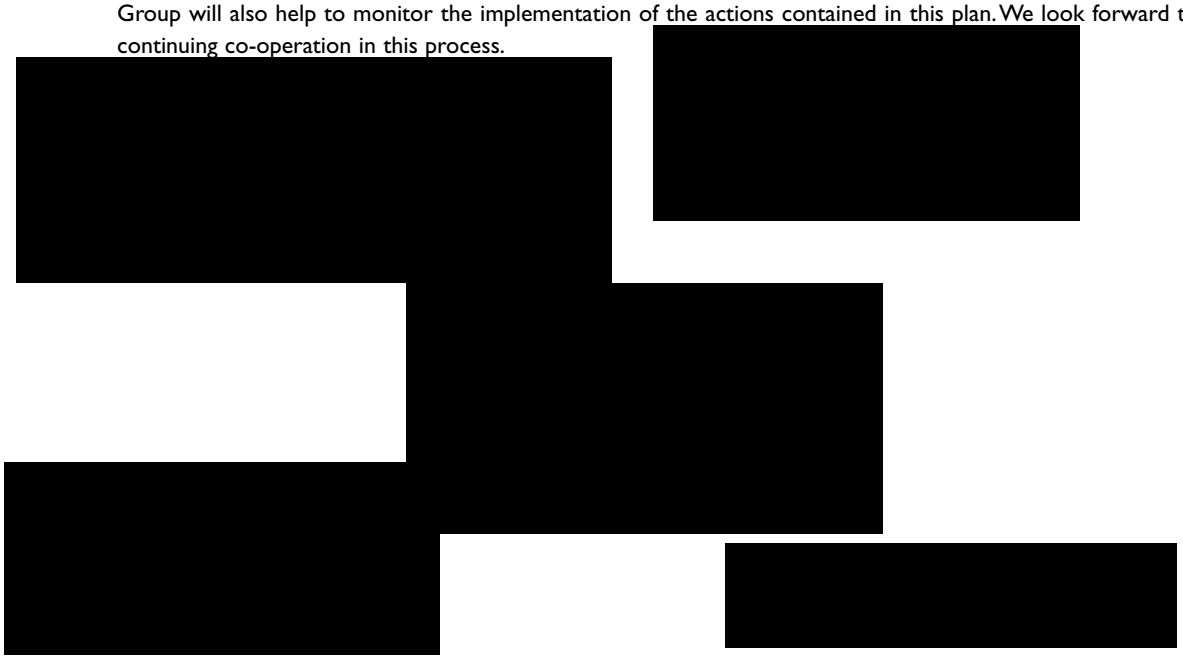
FOREWORD

In June 1992 the Prime Minister and over 150 other heads of state or governments signed the Convention on Biological Diversity at Rio de Janeiro. They did so to express a shared belief that action must be taken to halt the worldwide loss of animal and plant species and genetic resources. They recognised that each country has the primary responsibility to save and enhance biodiversity within its jurisdiction. At the same time they agreed to draw up national plans and programmes and to share resources to help implement them.

This document represents the first United Kingdom biodiversity action plan. It has been drawn up to a tight timetable, as the Prime Minister promised shortly after Rio, to demonstrate our commitment to the Convention. We accept that we have a contribution to make to the richness of the world's wildlife. The Darwin Initiative, the Aid Programme, the development of biodiversity action plans for our Dependent Territories and the fine work of our Institutions will help towards this aim. We also accept that we should further develop our strategies and programmes to ensure the conservation and, where possible, the enhancement of biodiversity within the UK. As the plan shows the Government and others in the UK are already strongly committed to the objectives of the convention, and are putting them into practice.

The plan which follows commits the Government, but just as we were not able to draw up the document without a wide and vigorous contribution from people and organisations who care about our natural heritage, so we shall not be able to deliver the plan without their active participation. The Government can take a lead and establish a framework but whether, in the end, we and our children enjoy a country which is richer or poorer in species and habitats depends on all of us.

We welcome debate and comments on the plan from home and abroad. We regard it as a first attempt to draw together a demanding programme of activity and commitment. To accelerate progress, and to sustain the open process by which this document has been prepared, the Government has agreed to set up a Biodiversity Action Plan Steering Group with membership drawn from central and local government, agencies, collections, academic bodies and non-governmental organisations. Its task will be to oversee the development of a range of targets for biodiversity so that they can be adopted in 1995 the European Nature Conservation Year. The Steering Group will also help to monitor the implementation of the actions contained in this plan. We look forward to continuing co-operation in this process.



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OUTLINE OF THE ACTION PLAN

SECTION 1

This section describes the United Kingdom's biological resource, and its importance in relation to Europe and the rest of the world. Chapter 1 explains the commitments made at the United Nations Conference on Environment and Development (the "Earth Summit"). It defines what is meant by biodiversity, why it is important and the objectives of the action plan. Chapter 2 examines the growth of our scientific knowledge, our scientific tradition and collections, and the role of scientific research in conserving our biodiversity. It sets out our commitment to continue this tradition by underpinning policy and programmes with sound science. Chapter 3 looks at UK biodiversity from a historical and geographical perspective. It describes the range and variation of species and habitats, and the genetic variation within species to be found in the UK, and why UK biodiversity is special and significant.

SECTION 2

This section describes the UK's strategy and programmes, and examines threats, problems, and opportunities. Chapter 4 looks at conservation within habitats (in-situ) and how the nature conservation agencies implement government strategy through the designation of protected areas and, increasingly, through measures to restore biodiversity in the wider countryside. Chapter 5 examines conservation outside natural habitats (ex-situ) and the techniques which are employed for conserving plants, animals and micro-organisms through collections, storage and propagation (among others). Chapter 6 is concerned with the sustainable use of those natural resources which contribute to biodiversity and how they may be conserved and enhanced for future generations. Chapter 7 explains the role of environmental awareness and education, and the contribution that can be made by all sectors of our society. Chapter 8 describes the support the UK gives to biodiversity overseas and our special responsibility for Antarctica, the Crown Dependencies and the Dependent Territories. Chapter 9 explains the importance of information and data and the need to establish a national biota database.

SECTION 3

The final section, chapter 10, draws the components of the action plan together, and provides a forward work programme. It describes the mechanisms for implementation, reporting and review, and proposes a new Biodiversity Action Plan Steering Group which will have responsibility for overseeing the development of targets for biodiversity, and the techniques and programmes necessary to achieve them. The outcome will be reported in 1995 which has been designated by the Council of Europe as the European Nature Conservation Year.

NOTES

This three stage approach is generally in line with the UN Environment Programme guidelines, of national assessment of country study/national strategy/action plan. Because of the UK's tradition and science base developed over 200 years, we have decided to include the country study, the strategy and the action plan in one document.

The adjective 'British' is used throughout the action plan to mean 'of the United Kingdom'.

The definition of biodiversity provided by Article 2 of the Biodiversity Convention is:

> *'The variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.'*

This plan addresses biodiversity in this sense with primary reference to the UK, but with a chapter on UK responsibilities overseas. It does not consider the issue of biotechnology which is also dealt with in the Biodiversity Convention, as this is covered in the parallel document *Sustainable Development: The UK Strategy*.

For more information about the UK Biodiversity Action Plan visit http://www.jncc.gov.uk/page-5155

UK Biodiversity Group

Tranche 2 Action Plans

Maritime Species and Habitats

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Coastal vegetated shingle Habitat Action Plan

1. Current status

1.1 Physical and biological status

1.1.1 Shingle is defined as sediment with particle sizes in the range 2-200 mm. It is a globally restricted coastal sediment type with few occurrences outside north-west Europe, Japan and New Zealand. Shingle beaches are widely distributed round the coast of the UK, where they develop in high energy environments. In England and Wales it is estimated that 30% of the coastline is fringed by shingle. However most of this length consists of simple fringing beaches within the reach of storm waves, where the shingle remains mobile and vegetation is restricted to temporary and mobile strandline communities.

1.1.2 Shingle structures take the form either of spits, barriers or barrier islands formed by longshore drift, or of cuspate forelands where a series of parallel ridges piles up against the coastline. Some shingle bars formed in early post-glacial times are now partly covered by sand dunes as a result of rising **s**ea levels leading to increased deposition of sand.

1.1.3 The origin of coastal shingle varies according to location. In southern England, much of it is composed of flint eroded out of chalk cliffs. Shingle deposits of Ice Age origin lying on the sea bed may be reworked by wave action and redeposited or moved by longshore drift along the coast. In northern and western Britain, shingle may derive from deposits transported to the coast by rivers or glacial outwash. Shingle structures are of geomorphological interest.

1.1.4 The vegetation communities of shingle features depend on the amount of finer materials mixed in with the shingle, and on the hydrological regime. The classic pioneer species on the seaward edge include sea kale *Crambe maritima*, sea pea, *Lathyrus japonicus,* Babington's orache*, Atriplex glabriuscula,* sea beet, *Beta vulgaris,* and sea campion *Silene uniflora*; such species can withstand exposure to salt spray and some degree of burial or erosion. Further from the shore, where conditions are more stable, more mixed communities develop, leading to mature grassland, lowland heath, moss and lichen communities, or even scrub. Some of these communities appear to be specific to shingle, and some are only known from Dungeness. On the parallel ridges of cuspate forelands, patterned vegetation develops, due to the differing particle size and hydrology. Some shingle sites contain natural hollows which develop wetland communities, and similar vegetation may develop as a result of gravel extraction.

1.1.5 Shingle structures may support breeding birds including gulls, waders and terns. Diverse invertebrate communities are found on coastal shingle, with some species restricted to shingle habitats.

1.1.6 Shingle structures sufficiently stable to support perennial vegetation are a comparatively rare feature even in the UK. The major vegetated shingle structures surveyed in 1987-1991 by Sneddon and Randall totalled some 5000 ha in England, 700 ha in Scotland and 100 ha in Wales. Dungeness, in southern England, is by far the largest site, with over 2000 ha of shingle, and there are only five other structures over 100 ha in extent in the UK. The main concentrations of vegetated shingle occur in East Anglia and on the English Channel coast, in north-east Scotland, and in north-west England and south-west Scotland. The Welsh coast has a number of small sites**.** This habitat is poorly represented in Northern Ireland, where the key site is Ballyquintin in County Down.

1.2 Links with species action plans

1.2.1 The following BAP priority species have significant populations on vegetated shingle sites: toadflax brocade *Calophasia lunula*, white spot *Hadena albimacula*, stinking hawk's-beard *Crepis foetida*, small-flowered catchfly *Silene gallica*, endemic sea lavenders *Limonium* spp, red hemp-nettle *Galeopsis angustifolia*, brown-banded carder bee *Bombus humilis*, large garden bumble bee *Bombus ruderatus*, short haired bumble bee *Bombus subterraneus*, and the hopper *Aphrodes duffieldi*. Wetlands within shingle sites are also important for the following species: medicinal leech *Hirudo medicinalis,* and great crested newt *Triturus cristatus*.

2. Current factors affecting the habitat

2.1 **Sediment supply**. The health and ongoing development of a shingle feature depend on a continuing supply of shingle. This may occur sporadically as a response to storm events rather than continuously. It is frequently lacking owing to interruption of coastal processes by coast defence structures, by offshore aggregate extraction or by artificial redistribution of material within the site (eg Dungeness). Attempts have been made to rectify the situation by mechanical reprofiling, which is likely to fail in the long run because it does not address the lack of new material, or by beach recharge.

2.2 **Natural mobility**. Shingle features are rarely stable in the long term. Many structures exhibit continuous

longshore drift, and ridges lying parallel to the shoreline tend to be rolled over towards the land by wave action in storm events. This movement has a knock-on effect on low-lying habitats behind the shingle. Movement is likely to be accelerated by climate change resulting in sea level rise and increased storminess.

2.3 **Exploitation**. Shingle structures have been regarded as a convenient source of aggregates, and have been subject to varying degrees of extraction resulting in severe alteration of morphology and vegetation (eg Dungeness and Spey Bay) or almost total destruction of major parts of the feature (eg Rye Harbour). Industrial plant, defence infrastructure and even housing have been built on shingle structures (eg Dungeness, Orfordness, Spey Bay), destroying vegetation and ridge morphology. At Dungeness water is abstracted from the groundwater system; there is some evidence of drought stress on the vegetation, but it is difficult to distinguish the effects of water abstraction from those of gravel extraction.

2.4 **Access**. Shingle vegetation is fragile; the wear and tear caused by access on foot, and particularly by vehicles, has damaged many sites. The causes include military use, vehicle access to beaches by fishermen, and recreational use. Such disturbance can also affect breeding birds.

2.5 **Grazing**. In a few cases areas of shingle were traditionally grazed, but this management has now largely ceased, leading to domination by willow carr on wetlands and changes to vegetation structure. The impacts of removal of grazing on breeding birds and other shingle species are not fully understood.

3. Current action

3.1 Legal status

3.1.1 Vegetated shingle is a rare habitat; all major examples and many minor ones have therefore been notified as SSSIs or ASSIs. Many are also declared as NNRs or LNRs, or are owned by voluntary conservation bodies. Vegetated shingle is listed as a habitat type under Annex I of the EC Habitats Directive ('Perennial vegetation of stony banks'), and five sites in England and two in Scotland are proposed as SACs. Three of the former have also been submitted or classified as SPAs under the EC Birds Directive.

3.2 Management, research and guidance

3.2.1 Shingle sites which are reserves and/or hold designations receive some protection from further damage, but many of them have been damaged in the past, and there is little positive management of the habitat. It is often impossible to control recreational use by third parties. The main exception is Orfordness, which was acquired by the National Trust in 1993 and is a candidate SAC. Here, EU LIFE funding has been obtained for rehabilitation of the site and experimental re-creation of the ridge system, and public access is being controlled. The management plan for the MoD's holding on Dungeness proposes positive measures for the re-establishment of vegetation.

3.2.2 A survey of the major vegetated shingle structures of Great Britain was commissioned by NCC in 1987. The results were published by JNCC in 1993 and 1994, and comprise a new classification of shingle vegetation and descriptions of all major and many minor vegetated shingle sites.

3.2.3 The UK Government has set out its commitment to sustainable management of the coast in a number of publications. These include DETR's (formerly DoE) *Policy Guidelines for the Coast* and *Planning Policy Guidance - Coastal Planning* (PPG 20), and SO's *Coastal Planning* (NPPG 13). A *Coastal Planning Technical Advice Note* has been prepared for Wales. DoE(NI)'s *Planning Strategy for Rural Northern Ireland* has provisions relating to development, access and conservation of the coast. MAFF and the Welsh Office have also produced a *Strategy for Flood and Coastal Defence in England and Wales* and DETR has produced *Coastal Zone Management - Towards Best Practice*. DETR's Coastal Forum was set up in 1994; similar fora have recently been initiated in Scotland and Wales, and one is expected shortly in Northern Ireland.

3.2.4 Environmental Impact Assessment is a statutory requirement for certain proposed developments where there is likely to be a significant effect on the environment.

4. Action plan objectives and proposed targets

Targets will depend on the results of research and testing.

4.1 Prevent further net loss of existing vegetated shingle structures totalling about 5800 ha. (However local gains and losses due to storm events occur sporadically and should be accepted provided that the national and regional resources are maintained overall.)

4.2 Prevent, where possible, further exploitation of, or damage to, existing vegetated shingle sites through human activities, and maintain the quality of existing plant and invertebrate communities which are currently in favourable condition.

4.3 Achieve the restoration, where possible, of degraded or damaged habitats of shingle structures, including landward transitions, where such damage has been extensive and natural recovery is not likely to be initiated, by 2010.

5. Proposed action with lead agencies

5.1 Policy and legislation

5.1.1 Permit as far as possible the natural movement of coastal sediments through coastal processes which maintain shingle structures in favourable condition, including the natural landward movement of shingle banks. (ACTION: CCW, DETR, DoE(NI), EN, MAFF, NAW, SE, SNH)

5.1.2 Continue the current presumption against the extraction of the sub-tidal shingle resources unless environmental and coastal impact concerns, including those affecting coastal shingle structures and future supply of material to shingle shorelines, can be satisfactorily resolved. Continue to seek opportunities for the revocation or surrender of existing licences where appropriate. (ACTION: CEC, DETR, DoE(NI), NAW, SE)

5.1.3 Subject applications to extract aggregate from coastal shingle structures to the most rigorous examination and, where appropriate, seek opportunities for the surrender or revocation of existing permissions. (ACTION: DETR, DoE(NI), LAs, NAW, SE)

5.1.4 Ensure that the importance of shingle structures and offshore shingle resources is recognised in flood and coastal defence strategies and, where appropriate, encourage such strategies to contribute to the objectives and targets of this plan. (ACTION: DANI, DoE(NI), EA, MAFF, NAW, SE)

5.2 Site safeguard and management

5.2.1 Apply conservation designations (including NNR status where appropriate) to remaining areas of shingle which meet national or international criteria for site selection, and ensure appropriate management of designated sites by 2004. (ACTION: CCW, EHS, EN, SNH)

5.2.2 Negotiate positive management agreements on vegetated shingle SSSIs and ASSIs. (ACTION: CCW, EHS, EN, SNH)

5.2.3 If pilot projects (see 5.5.4) are successful, promote the application of techniques, where appropriate, in a wider programme of rehabilitation on major shingle sites suffering from gross and extensive damage. (ACTION: CCW, EHS, EN, MoD, SNH)

5.2.4 Encourage reinstatement of wetland vegetation on shingle sites (where appropriate) by scrub clearance and grazing. (ACTION: EN)

5.3 Advisory

5.3.1 Promote and develop demonstration sites for the management and rehabilitation of shingle structures and disseminate best practice. (ACTION: CCW, EHS, EN, SNH)

5.3.2 Ensure all relevant agri-environment project officers and members of regional agri-environment groups are advised of the location of existing examples of this habitat, its importance and the management requirements for its conservation. (ACTION: CCW, EN, SNH)

5.3.3 Allowing natural landward movement of shingle features (see 5.1) will, in some cases, affect other habitats such as saline lagoons, grazing marsh, fens and reedbeds, some of which will be designated sites. The implementation groups for the relevant HAPs should be advised on how to make appropriate provision for habitat creation. In some cases, breaches in shingle banks may lead to the development of saltmarsh habitats and this needs to be taken account of in the respective HAPs. (ACTION: CCW, EA, EN, SNH)

5.4 International

5.4.1 Develop international links to promote the exchange of information and development of best practice in the management of coastal vegetated shingle. (ACTION: JNCC)

5.5 Monitoring and research

5.5.1 Assess the current extent of damage to shingle vegetation and geomorphology in order to inform the setting of restoration targets within the first three years of this plan. (ACTION: CCW, EHS, EN, SNH)

5.5.2 Collate and disseminate information on changes in the extent and quality of the vegetated shingle resource in the UK in order to enable effective monitoring of the objectives of this plan. (ACTION: CCW, EHS, EN, JNCC, SNH)

5.5.3 Continue research into the use of remote sensing for monitoring soft coast habitats including shingle structures. (ACTION: EA)

5.5.4 Carry out pilot projects to test the methods for the practical restoration of damaged shingle structures and their vegetation and morphology, and disseminate the results. (ACTION: EN)

5.5.5 Initiate research to determine the relationship between offshore shingle banks and onshore shingle structures in relation to aggregate extraction. (ACTION: CEC, DETR, DoE(NI), MAFF, NAW, SE)

5.5.6 Initiate research to assess the likely medium to long-term demand for offshore shingle in order to maintain the current vegetated shingle structures and their associated shingle shorelines. (ACTION: CEC, DETR, DoE(NI), MAFF, NAW, SE)

5.6 Communications and publicity

5.6.1 Increase public awareness of the value and fragility of vegetated shingle through on-site interpretation. (ACTION: CCW, EHS, EN, SNH)

6. Costings

6.1 The successful implementation of the habitat action plans will have resource implications for both the private and public sectors. The data in the table below provide an estimate of the likely additional resource costs to the public and private sectors. These additional resource costs are based on the annual average over 5 and 10 years. The total expenditure for these periods of time is also given. Three-quarters of the additional resources are likely to fall to the public sector.

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Costings for vegetated shingle

	Current expenditure	1st 5 yrs to 2004/2005	Next 10 yrs to 2014/2015
Current expenditure /£000/Yr	0		
Total average annual cost /£000/Yr		102.5	128.7
Total expenditure to 2005/£000		512.3	
Total expenditure 2005 to 2014/£000			1287.3

* There is little or no current payment for shingle habitats under agri-environment schemes or other positive management payments.

Making Space for Nature:
A review of England's Wildlife Sites and Ecological Network

Chaired by Professor Sir John Lawton CBE FRS

Submitted to the Secretary of State, the Department for Environment, Food and Rural Affairs on 16 September 2010

Executive Summary

Do England's wildlife sites comprise a coherent and resilient ecological network? If not, what needs to be done? These are the questions that we aim to answer in this report. We first consider why these questions are important in the context of past, current and future pressures on the environment, and describe what ecological networks are and the benefits they bring. We go on to consider the strengths and weaknesses of our current wildlife sites, before setting out a prioritised set of ecological solutions to improve the network. Finally, we make 24 recommendations for practical action to *Make Space for Nature* and achieve a coherent and resilient ecological network.

Throughout, we stress that what is needed is a step-change in nature conservation. We need to embrace a new, restorative approach which rebuilds nature and creates a more resilient natural environment for the benefit of wildlife and ourselves. This will require strong leadership from government, but is not a job for government alone. It will require effective and positive engagement with the landowners and land managers. And it will need improved collaboration between local authorities, local communities, statutory agencies, the voluntary and private sectors, farmers, other land-managers and individual citizens.

Why is having a coherent and resilient ecological network important?

England's wildlife and landscapes have inspired and delighted through generations. There are strong moral arguments for recognising the intrinsic values of other species and for passing on the natural riches we have inherited to future generations. We have also recently begun to better understand (or perhaps remember) that our natural world is not a luxury: it is fundamental to our well-being, health and economy. The natural environment provides us with a range of benefits – *ecosystem services* including food, water, materials, flood defences and carbon sequestration – and biodiversity underpins most, if not all, of them. The pressures on our land and water are likely to continue to increase and we need to learn how to manage these resources in ways which deliver multiple benefits, for example, achieving profitable and productive farming while also adopting practices which enhance carbon storage, improve flood water management and support wildlife.

Declines in wildlife are a global problem: the World has failed to meet its commitment to achieve a significant reduction in the rate of global biodiversity loss by 2010; Europe has not met its 2010 target to halt biodiversity loss. The report summarises the losses that have occurred, and continue to occur, in England's wildlife. Across species groups we have seen significant declines, in particular since the end of the Second World War, mainly as a result of changes in land use which has led to the loss and deterioration of many wildlife habitats. But it is not all bad news: for some habitats and species the historic patterns of losses have slowed, and in some cases they have been reversed (often through the conservation efforts of statutory bodies, the voluntary sector and landowners). Nonetheless, across many groups of species it is the more 'specialist' species that tend to be in decline while the less choosy, more adaptable 'generalists' tend to be faring better - an indication of ongoing declines in the quality and variety of England's natural environment.

Concern for nature is not new, and considerable progress has been made over the last century to protect wildlife. Voluntary conservation organisations helped to stimulate popular

and Government support, culminating in the National Park and Access to the Countryside Act 1949 which laid the foundations for designating places that are special for wildlife (notably National Nature Reserves and Sites of Special Scientific Interest, SSSIs) and people (National Parks and Areas of Outstanding Natural Beauty). Subsequent legislation has improved first the protection, and more recently the management, of wildlife sites in particular SSSIs. Despite the important contribution designated sites have made, England's wildlife habitats have become increasing fragmented and isolated, leading to declines in the provision of some ecosystem services, and losses to species populations.

Ecological networks have become widely recognised as an effective response to conserve wildlife in environments that have become fragmented by human activities. An ecological network comprises a suite of high quality sites which collectively contain the diversity and area of habitat that are needed to support species and which have ecological connections between them that enable species, or at least their genes, to move. Over 250 ecological networks are being planned or implemented at international, national and regional levels around the world, including in some parts of the UK. Provision for ecological networks is made in a number of international treaties and agreements but England has not yet met its commitments under these agreements. Taking steps to do so, however, will deliver a range of benefits for people as well as wildlife, because of the range of ecosystem services that resilient, coherent ecological networks can provide.

We propose that the overarching aim for England's ecological network should be to deliver a natural environment where: *Compared to the situation in 2000, biodiversity is enhanced and the diversity, functioning and resilience of ecosystems re-established in a network of spaces for nature that can sustain these levels into the future, even given continuing environmental change and human pressures.*

We also recommend that this be underpinned by three objectives:

(1) To restore species and habitats appropriate to England's physical and geographical context to levels that are sustainable in a changing climate, and enhanced in comparison with those in 2000.
(2) To restore and secure the long-term sustainability of the ecological and physical processes that underpin the way ecosystems work, thereby enhancing the capacity of our natural environment to provide ecosystem services such as clean water, climate regulation and crop pollination, as well as providing habitats for wildlife.
(3) To provide accessible natural environments rich in wildlife for people to enjoy and experience.

Future Challenges

Demographic change, economic growth, new technologies, societal preferences and changes in policy and regulatory environments (international, EU and domestic) may all have profound consequences - some potentially positive - for our ability to establish a coherent and resilient ecological network. Climate change, particularly in the longer term, may have the biggest impact of all. We are already seeing a number of changes as a result of climate: shifts in species ranges; changes in the timing of seasonal events (with emerging mismatches in the timing of events for currently inter-dependent species); and habitat preferences altering. Not all changes will be harmful, for example, many of England's

southern species may be able to increase their range by expanding northwards. In the longer term, however, species may struggle to survive, and other impacts such as sea-level rise, an increase in extreme weather events, and other changes to ecosystem processes (e.g. caused by summer droughts) are likely to have further profound, and largely negative, effects.

Establishing a coherent and resilient ecological network will help wildlife to cope with these changes. It will also improve the ability of our natural environment to provide a range of high quality ecosystem services today and in the future. It can help us both mitigate and adapt to climate change by, for example, storing carbon or improving the security of water supplies. We don't know exactly what the future holds, and we don't know exactly which species will be the winners and losers. Conserving the ecosystems and wildlife that we have today will, however, maximise our future options and improve our chances of achieving health and prosperity for ourselves and our children.

The Nature and Status of our current Wildlife Sites and Ecological Network

England has a wide range of different types of statutory and non-statutory sites which, to a greater or lesser extent, support our wildlife. We take a broad view of what comprises a 'wildlife site' and identify three general categories ('Tiers'), which include eleven types of sites. Tier 1 sites are those whose primary purpose is nature conservation and which have a high level of protection (e.g. SSSIs); Tier 2 sites are designated for their high biodiversity value but do not receive full protection (e.g. Local Wildlife Sites); Tier 3 are landscape designations with wildlife conservation as part of their statutory purpose (National Parks and AONBs).

We examine the extent to which these different Tiers of sites separately and collectively comprise a coherent and resilient ecological network by testing the evidence against five attributes that we identify for such a network:

(i) The network will support the full range of England's biodiversity and incorporate ecologically important areas, including special biodiversity.
(ii) The network and its component sites will be of adequate size, taking account of the needs of our natural environment to adapt to climate change.
(iii) The network sites will receive long-term protection and appropriate management.
(iv) Sufficient ecological connections will exist between sites to enable species movement.
(v) Sites will be valued by, and be accessible to people, including sites close to where they live.

We draw upon a wide range of evidence including academic literature, organisational reports, evidence submitted to the panel by more than 45 organisations and novel analyses carried out specifically for this review.

The evidence demonstrates that the SSSI series, as important as it is, clearly does not in itself comprise a coherent and resilient ecological network. Perhaps this should not come as a surprise since SSSIs were not designated with this aim in mind. Looking across all three Tiers of wildlife sites, the evidence demonstrates that only attribute (i) is substantially met; in all other cases there are serious short-comings in the network. Notably, many of England's

wildlife sites are too small; losses of certain habitats have been so great that the area remaining is no longer enough to halt additional biodiversity losses without concerted efforts; with the exception of Natura 2000 sites and SSSIs, most of England's semi-natural habitats important for wildlife are generally insufficiently protected and under-managed; many of the natural connections in our countryside have been degraded or lost, leading to isolation of sites; and too few people have easy access to wildlife.

Many species are now largely restricted to wildlife sites simply because they have mostly been lost from everywhere else. We need to take steps to rebuild nature.

Rebuilding Nature

The essence of what needs to be done to enhance the resilience and coherence of England's ecological network can be summarised in four words: *more*, *bigger*, *better* and *joined*. There are five key approaches which encompass these, and also take account of the land around the ecological network. We need to:

(i) Improve the quality of current sites by better habitat management.
(ii) Increase the size of current wildlife sites.
(iii) Enhance connections between, or join up, sites, either through physical corridors, or through 'stepping stones'.
(iv) Create new sites.
(v) Reduce the pressures on wildlife by improving the wider environment, including through buffering wildlife sites.

We set out an ecological rationale for deciding between these approaches. In general, the first priority is to enhance the quality of remaining wildlife habitat. Increasing connectivity helps, but first there needs to be high quality sites with thriving wildlife populations to connect. Local circumstances are also important. In areas which have large amounts of relatively unfragmented habitat, the best strategy will often be to focus on improving management and enhancing habitat diversity ('heterogeneity'). In contrast, in areas which only have small and isolated sites, it will be better to invest in the restoration and creation of new wildlife habitat.

Importantly, there are trade-offs between these actions: the more we do to improve the quality of existing sites or to reduce the pressures on them by enhancing the wider environment, the less we will need to do to create new wildlife sites.

Establishing a coherent and resilient ecological network

The approaches required to achieve a coherent and resilient ecological network are varied, and we make 24 wide-ranging recommendations. Five themes unite them:

(i) We need to continue the recent progress in improving the management and condition of wildlife sites, particularly our SSSIs. We also make recommendations for how these should be designated and managed in ways that enhance their resilience to climate change.
(ii) We need to properly plan ecological networks, including restoration areas. Restoration needs to take place throughout England. However, in some areas, both the scale of what can be delivered to enhance the network, and the ensuing

(iii) There are a large number of surviving patches of important wildlife habitat scattered across England outside of SSSIs, for example in Local Wildlife Sites. We need to take steps to improve the protection and management of these remaining wildlife habitats. 'Protection' will usually be best achieved through incentive-based mechanisms, but at times may require designation.

(iv) We need to become better at deriving multiple benefits from the ways we use and interact with our environment. There are many things that society has to do that may seem to have rather little to do with nature conservation, but could have, or even should have if we embrace more radical thinking; flood management by creating wetlands is an obvious example. We need to exploit these 'win-win' opportunities to the full. Being better at valuing a wider range of ecosystem services would help this process.

(v) We will not achieve a step-change in nature conservation in England without society accepting it to be necessary, desirable, and achievable. This will require strong leadership from government and significant improvements in collaboration between local authorities, local communities, statutory agencies, the voluntary and private sectors, farmers, landowners and other land-managers and individual citizens

Our recommendations set out a 'direction of travel'. They do not provide a detailed, prescriptive plan of exactly what should happen, where. The precise solutions need to take full account of local opportunities and ambition, although we believe that some form of national framework is required to ensure coherence and cost-efficiencies across the network (what happens in one place will affect what is needed elsewhere).

Partly because we have not set out a detailed, prescriptive plan, and partly because many of the recommendations overlap and interact (i.e. what we have to do to implement one recommendation will often depend upon how successful we are in implementing others), it has not been possible to provide accurate estimates of the cost of each of the 24 recommendations. Nonetheless, taking account of recent comprehensive biodiversity cost-estimates derived by other studies, and including current expenditure, we estimate that the total annual costs of establishing a coherent and resilient network will be in the range of £600 million to £1.1 billion.

The next few years are clearly going to be a time of budgetary constraint, when additional resources are unlikely to be available. We recognise that it will not be possible to take all necessary action immediately, or even soon. We do, however, need to plan for the medium and longer term and *Make Space for Nature*. Amongst this uncertainty, there is one thing of which we can be certain: the sooner we act to establish a coherent and resilient ecological network, the lower the eventual cost and the greater the benefits for us all.

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Box 9. Canals create natural connections for wildlife and people

The canal network is one of the most accessible and well-used facilities for recreation in the country. It is not often appreciated that 96% of a waterways' use is on the land, not the water. A key strength of the waterway network is its multi-functionality and the links it creates between urban areas and the wider countryside. Canals and rivers also comprise high quality wildlife habitats, used by a wide range of species, including many that are rare or protected. British Waterways' network in England includes over 54 SSSIs, many of which are canal or navigable river.

In 2009 there were 346 million visits to waterway towpaths in Great Britain, of which over 80% were for leisure or recreation. Surveys of towpath visitors indicate that 62% consider that the presence of the waterway towpath helps increase the amount of physical activity they regularly take. Improving just a single kilometre of canal towpath will, typically, attract an additional 40,000 visits, generate a further £50,000 of visitor spend for local communities (securing 1.3 jobs), deliver quality of life benefits of £20,000 and up to £18,000 of additional health benefits, and save around 100 tonnes of CO_2 every year

Example - Droitwich Canals Restoration
The restoration of the Droitwich Canals will open up a 21 mile ring of cruising waterways in Worcestershire. The environmental benefits associated with the restoration include the creation of 5 ha of reedbed and the regeneration of 24 ha of biodiversity-rich brownfield land. The restoration will create a continuous wildlife corridor enabling priority species such as water vole, otter and bats to forage and disperse further afield. It is estimated that the completion of the restoration will result in 322,000 additional visitor days each year, 12000 cycling visits, 2000 canoe visits, 3500 angling visits and the creation of 196 full-time jobs. This is likely to contribute an additional £2.75 million to the local economy and an increase in the value of adjacent housing of over £3 million.

(Source: British Waterways)

6.4.3 Biodiversity offsetting and developer contributions

Biodiversity offsets established through the planning process are another mechanism that could be used to enhance ecological networks. Offsets are activities designed to compensate for residual and unavoidable harm to existing wildlife sites caused by development activity. We discuss the idea in some detail because it is not something that we have dealt with elsewhere in this report.

In England, mandatory offsetting is currently only required where development of overriding public interest has significant impacts on the Natura 2000 network or sites occupied by European protected species. Planning policy encourages, but does not absolutely require, local authorities to ensure compensation for development impacts on biodiversity (e.g. PPS9 through s106 agreements). Based on the evidence of developer contributions in England and other biodiversity offsetting schemes from around the world, including the use of 'conservation credits' and habitat banking in other countries, there are a number of principles that must underpin an effective system:

(i) Biodiversity offsetting must not become a 'licence to destroy' or damage existing habitat of recognised value. In other words, offsets must only be used to compensate for genuinely unavoidable damage. Development should avoid

adverse impacts first, mitigate impacts second and compensate for unavoidable impacts as a last resort.

(ii) Where developers propose to create replacement habitat there needs to be some certainty that the habitat type can be (re-)created. Applying the precautionary principle, and recognising that some habitat creation schemes may be less successful than initially planned, each individual offset scheme should aim to achieve a net gain for biodiversity.

(iii) Benefits should be secured in the long term with supporting mechanisms in place to deliver long term management (often more than 25 years).

(iv) There must be recognition that some habitats cannot be re-created (e.g. ancient woodland) while others can take decades to develop their wildlife interest.

(v) 'Receptor areas' for creating habitat must not be places of existing high wildlife value.

(vi) Suitable multiplier ratios need to be applied to compensation in recognition that the new site may be of a different value to the network than the original one, and to take account of factors such as distance from the site of the damage, the time needed for habitat creation, the types of habitat being lost and accessibility for people. Usually, these multipliers will be greater than one, but they need not always be.

(vii) Wherever possible, the created habitat should be in place before the original site is lost.

(viii) Offsets should be used to compensate both for the loss of current wildlife sites and for damage to other wildlife network components, for example areas identified as important for ecological restoration or connectivity but not yet managed as such.

(ix) Opportunities should be taken to pool habitat compensation from different developments so that larger habitat blocks can be created. There should be community agreement on what is to be achieved (and what contributions are expected) through local plans, so developers are clear at the outset what scale of contribution is expected from them.

We conclude that in the context of establishing and maintaining ecological networks:

a) There are risks that biodiversity offsetting could undermine ecological networks if they lead to any reduction in the levels of protection afforded to wildlife sites and habitats. It may be possible to mitigate these risks by ensuring that a system of biodiversity offsets is underpinned by a clear set of principles, as proposed above.

b) A well-managed scheme can bring benefits to the ecological network by effectively pooling a number of offsets required for separate small developments into a larger and more beneficial habitat block. This can be done without imposing additional burdens on developers.

c) The operation of a system of biodiversity offsets could deliver net gains for wildlife by providing an opportunity for developers (or other interested organisations) to buy additional *conservation credits* as part of their social responsibility commitments. The financial value of the credits could also

reflect the value of other ecosystem services such as carbon storage (Section 6.3.4)

d) A responsible authority needs to be identified to oversee the administration of biodiversity offset schemes and verify effective offset delivery.

e) We need to further develop the evidence base in a number of important areas including:
   - refining creation and restoration techniques for certain habitats;
   - establishing the appropriate multipliers needed to ensure full compensation, and developing rules for offsetting 'out of kind' (where damage to one type of habitat is compensated for by providing another);
   - verifying the market mechanisms through which biodiversity offsets can operate effectively in an English context.

We suggest that the evidence base is developed, and the overarching principles tested, through pilot schemes in willing local authority areas.

Recommendation 22. If a formal system of biodiversity offsets is to be introduced, pilot schemes should be established to test and refine its operation, to ensure it meets the conditions we have set out for a safe and effective system.

6.5 Improving the countryside

The ways in which the wider countryside outside the ecological network – the matrix – is managed is important. If the matrix is hostile, protected sites need to be large, but even small sites can support thriving wildlife if the land around them is sufficiently benign. A richer, less hostile environment around the network will also, in itself, provide space for nature and support the provision of ecosystem services at landscape scales.

One approach to reducing pressures is to establish buffers around specific sites (Section 2.1.3) and we have already identified the need for a new type of incentive measure for this purpose (Section 6.3.3)). Because this report is focussed on the network of wildlife sites, we have not provided a comprehensive assessment of ways to enhance the wider environment. Instead, we briefly consider the importance of voluntary incentives, before finally highlighting a number of other measures to reduce pressures on the network.

6.5.1 Entry Level Stewardship

Entry Level Stewardship (ELS) is designed to be a simple, multi-objective, flexible scheme, open to all agricultural land managers in England. It is the companion to the Higher Level Scheme (see section 6.3.3) and within it land managers are free to select any combination of options they feel suits their farming operation. Options are designed to be deliverable without advisory input or targeting, and as such are intended to be 'light touch' and to have minimal impacts on conventional and productive farming operations. The scheme is capable of delivering the requirements of some species with dispersed populations, able to improve the permeability of the wider countryside for some species, and to reduce inputs around priority

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Upland raptors and the assessment of wind farm impacts

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Government targets on renewable energy coupled with anthropogenic constraints on development have resulted in a surge in proposals to locate wind farms in upland areas, where they may conflict with the wellbeing of scarce or rare bird species including raptors. European and UK legislation demand that the effects of wind farm developments, both individually and in combination, be assessed to determine the level of impact on these species. The principle adverse effects of wind farms on raptors, as for other terrestrial birds, potentially involve disturbance (displacement or barrier impacts) or collision fatality. Few long-term studies on such effects of wind farms have been undertaken. We review available research results on displacement of raptors, which primarily involve foraging birds, and conclude that most studies indicate that displacement appears to be negligible, although some notable exceptions occur and more research is needed. There is also a need for better understanding of the numbers of birds likely to be killed through collision with turbine rotors at the site level in order to inform planning decisions, although models of bird distribution at several spatial scales can be used to circumvent potential difficulties when locating turbines. Modelling approaches have also been developed that attempt to quantify the theoretical risk of collision. One such approach, the Band model, is a valuable tool for impact assessment and its use is now widespread in the UK. However, there are practical problems associated with gathering the data required to run the model and numerous assumptions must be made concerning bird behaviour. This can lead to deficiencies in the input parameters which potentially have a large effect on the model outputs. Hence, we make recommendations for potential improvements, such as quantifying error in flight height estimation, training of observers in acuity skills, quantifying bird detection-distance functions, and research on factors influencing activity budgets and flight behaviour. In addition, the model outputs are usually adjusted to take account of turbine avoidance by birds and this aspect of birds' behaviour is poorly understood. As a result of these limitations, collision predictions are only indicative, and more reliable in some situations, and for some species, than others.

INTRODUCTION

A rapid increase in the number of wind farms in the UK is currently in progress (Drewitt & Langston 2006). The constraints on locating terrestrial wind farm developments are many and varied, including adequacy of the wind resource, concerns over impacts on the visual amenity of the landscape, and issues related to civil and military aviation, electro-magnetic transmissions, hydrology, ecology and archaeology. Crucially, locations in areas of relatively low conservation value are often otherwise contentious because of their close proximity to residential property. These factors have inevitably favoured proposals within upland areas that have low human population (SNH 2004), where they may conflict with the conservation of scarce and dispersed birds, including several species of raptors (Ratcliffe 1990).

European and UK legislation require that the impacts on nature conservation interests due to wind farm proposals, individually and in combination, are formally assessed under planning and environmental impact assessment procedures (e.g. Langston & Pullan 2003). However, we currently have limited understanding of the implications of the various potential

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effects of wind farms on birds (Drewitt & Langston 2006) and as a result there is a tendency for planning decisions to be heavily dependent on subjective judgement. Greater objectivity can only be achieved with better information on the factors influencing wind farm impacts and with better assessment tools to increase predictive capacity. In this paper we first review briefly studies that have aimed to investigate the principal factors affecting the impact of wind farm developments on upland raptors. We next discuss some of the predictive approaches being used to better quantify the likely effects of wind farms, especially regarding bird collisions with turbines. Finally, we draw attention to the many potential pitfalls in making quantitative estimates of collision risk and make suggestions for how methods may be improved.

EFFECTS OF WIND FARMS ON RAPTORS

The effects of a wind farm on birds depend on a wide range of factors including the specification of the development, topography and habitat of the surrounding land, and species present (Johnson *et al.* 2000a,b, Percival 2000, Erickson *et al.* 2002, Langston & Pullan 2003, Barrios & Rodriguez 2004, Smallwood & Thelander 2004, Hoover & Morrison 2005). Ultimately, the main effects of terrestrial wind farms on birds are determined by the sensitivity of individuals to disturbance and therefore whether displacement or a barrier effect (if disturbance occurs) or risk of collision with rotating turbine blades (if disturbance does not occur) is the primary impact. Disturbance can occur as a result of the operation of turbine rotors, human and machine noise during site construction activities, maintenance and repair work, and increased access (for agricultural or recreational purposes) following the establishment of site access tracks (Gill *et al.* 1996, Percival 2000, Langston & Pullan 2003, M. Reichenbach *pers. comm.*). Disturbance (visual and noise) may displace birds into less suitable habitat and this may reduce their ability to survive and reproduce (i.e. an indirect form of habitat loss through perceived predation risk) (Frid & Dill 2002). If not physically displaced, birds' foraging success may be impaired and other important life history behaviours disrupted (i.e. a form of habitat degradation) (F. Bergen *pers. comm.*). Alternatively, birds that are not displaced risk collision with rotating rotor blades (e.g. Erickson *et al.* 2001).

Most impact assessments to date have focused on displacement and collision risks and ignored the potential effects of behavioural disruption (Johnson *et al.* 2000a, F. Bergen *pers. comm.*). Clearly, birds cannot be simultaneously vulnerable to disturbance and collision since they are mutually exclusive spatially. However the relationship between the two effects may be unstable over time as birds may habituate to the presence of the wind farm (although see Stewart *et al.* 2005) or site faithful individuals may be lost from the population and not replaced. Post-construction impact studies should therefore ideally be conducted over a period relevant to the generation time of the species involved.

Displacement

Broadly, two methods have been used to investigate bird displacement effects at wind farms (Anderson *et al.* 1999): BACI (Before-After-Control-Impact) studies and those lacking a pre-construction contrast where abundance and distribution are contrasted between an operational wind farm site and a reference site lacking a wind farm. BACI methods provide the most powerful evidence but are probably less common, especially in European studies (Gill *et al.* 1996, Langston & Pullan 2003, M. Reichenbach *pers. comm.*). Displacement of breeding raptors may occur by disturbance preventing use of nest sites, and/or by disturbance preventing use of areas for other activities, notably foraging. Due to the low nesting density of breeding raptors (Newton 1979), studies involving potential displacement of raptors from nest sites are far less common than those examining displacement from foraging areas and inevitably involve lower sample sizes.

Potential displacement of raptor foraging has been investigated in several species, and at several wind farms for some species (Table 1). Although clearly more studies are highly desirable and more should be peer-reviewed in the public domain (Whitfield & Coupar, 2006), in general most research tends to show that disturbance of raptors at wind farms is negligible. Interesting differences are evident in some examples, however.

Two studies involving the Golden Eagle *Aquila chrysaetos* in the US have found no evidence of displacement, one being a simple comparison of the wind farm and a reference site (Schmidt *et al.* 2003) and a second BACI study (Johnson *et al.* 2000a). From data at a third US wind farm (the Altamont Pass Wind Resource Area in California) presented by Hunt *et al.* (1995) there was indication of some displacement by comparison with a reference area

Table 1. Summary of the results of several studies which have investigated potential displacement effects on raptors at wind farms.

Species	Sensitivity to displacement	Number of studies	References*
Turkey Vulture *Cathartes aura*	Low?	2	1, 4
Griffon Vulture *Gyps fulvus*	Low?	1	2
Red-tailed Hawk *Buteo jamaicensis*	Low	4	1, 3, 4, 5
Ferruginous Hawk *Buteo regalis*	Low?	2	1, 4
Rough-legged Buzzard *Buteo lagopus*	Low?	2	1, 3
Swainson's Hawk *Buteo swainsoni*	Low?	2	3, 4
Broad-winged Hawk *Buteo platypterus*	Low?	2	3, 5
Common Buzzard *Buteo buteo*	Low-Medium	6	6, 9, 10
Sharp-shinned Hawk *Accipiter striatus*	Low?	1	5
Golden Eagle *Aquila chrysaetos*	Low-High?	4	1, 4, 7, 8
Short-toed Eagle *Circaetus gallicus*	Low?	1	2
Common Kestrel *Falco tinnunculus*	Low	5	2, 6, 9, 10
American Kestrel *Falco sparverius*	Low	4	1, 3, 4, 5
Prairie Falcon *Falco mexicanus*	Low?	2	1, 4
Peregrine Falcon *Falco peregrinus*	Low?	2	1, 6
Red Kite *Milvus milvus*	Low?	3	6, 9, 10
Black Kite *Milvus migrans*	Low?	1	2
Hen Harrier *Circus cyaneus*	Low-Medium?	5	1, 3, 4, 5, 10
Montagu's Harrier *Circus pygargus*	Low-Medium?	2	9, 11
Marsh Harrier *Circus aeroginosus*	Low?	3	9, 10

1 = Schmidt *et al.* (2003); 2 = de Lucas *et al.* (2004); 3 = Johnson *et al.* (2000b); 4 = Johnson *et al.* (2004a); 5 = Kerlinger (2002); 6 = Phillips (1994); 7 = Hunt *et al.* (1995); 8 = Walker *et al.* (2005); 9 = F. Bergen (*pers. comm.*); 10 = M. Reichenbach (*pers. comm.*); 11 = M. de Lucas (pers.comm.)

The review refers to displacement from foraging areas, and not from nest sites, for which too few studies were available. Sensitivity to displacement classed as: Low = no evidence of any displacement effects, Medium = evidence of possible displacement in an area around individual turbines or from parts of a wind farm, High = evidence of complete displacement from a wind farm. A question mark indicates that it was difficult to reach a definitive conclusion because either: the study results were inconclusive or the number of studies was too low and/or studies did not involve a BACI protocol, or that interstudy variation in results clearly warranted further investigations.

away from the wind farm. Control of eagle prey in the wind farm may have produced this effect according to Stewart *et al.* (2005), although prey control was not encouraged by turbine owners until 1997 (Smallwood & Spiegel 2005), and the wind farm otherwise appeared to increase prey abundance (Smallwood & Thelander 2004). A long-term trend towards increasing raptor use of the Altamont Wind Farm may indicate that any earlier displacement no longer occurs (Smallwood & Thelander 2004). A study at a Scottish wind farm in Argyll (Walker *et al.* 2005) described shifts in the ranging behaviour of territorial Golden Eagles following wind farm construction and these shifts were consistent with avoidance of the entire wind farm area. The only flights observed crossing the wind farm related to nonterritorial birds, or excursions by the resident birds to intercept and aggressively interact with such birds. No change in the size of the range used by the eagles was apparent. However, the confounding effects of habitat change, a common problem in post-construction studies, must be acknowledged. Immediately prior to wind farm construction, 280 ha of plantation forest was cleared in order to mitigate the potential displacement effects. This resulted in a substantially increased foraging area located away from the wind farm.

At several wind farms, there have been no indications of Hen Harrier (*Circus cyaneus*) displacement, (Johnson *et al.* 2000b, Kerlinger 2002, M. Reichenbach *pers. comm.*, Schmidt *et al.* 2003) however Johnson *et al.* (2000a) found evidence of both small scale (< 100 m from turbines) and larger scale avoidance of turbines by harriers in the year following wind farm operation. Preliminary results at Argyll and Northern Ireland sites suggest foraging may be little affected, but local displacement of nesting attempts may occur in the order of 2–300 m around turbines (Natural Research unpubl. data).

Collision risk

Few wind farm collision casualties have been recorded in the UK, (e.g. Percival 2000). However, it is important to recognize that, to date, not many wind farms have been built in areas where high concentrations of activity make collisions likely.

Moreover, the data available thus far are mostly based on casually found victims, with no corrections for corpses that are overlooked or removed by scavengers (Langston & Pullan 2003); hence the studies can not be considered representative (Whitfield & Coupar 2006). Smallwood & Thelander (2004) also point out that several years of study may be necessary before reliable results are obtained. Remote methods of monitoring collision mortality may be the best way forward (Desholm 2003), although these need to be reliable and sufficiently inexpensive to allow installation at a large number of turbines to counter the likelihood that risk can vary substantially between turbines within a wind farm (Smallwood & Thelander 2004).

Better studies are available from abroad, although even here relatively few have countered all the potential pitfalls in methods used to document fatality rates (Whitfield & Coupar in press). Even when such potential shortcomings have been accounted for, simply presenting mortality rates per turbine or per installed MW, in the absence of any contextual information on the abundance of birds 'available' to be killed or otherwise at risk of death, does little to inform about the collision risk posed by wind farms (Smallwood & Thelander 2004). Low collision mortality may often be because wind farms have been located to avoid areas where bird concentrations are high or because turbines have been microsited to avoid localized areas of heavy use (e.g. Johnson *et al.* 2000a).

Despite such difficulties, there is a degree of consensus that raptors may be more vulnerable to collision than several other bird groups (NWCC 2000). This may be due to generally lower displacement or avoidance effects[1], although several other factors have been postulated as influential. The relatively greater fatalities of predatory species compared with scavengers such as the Turkey Vulture *Cathartes aura* or Raven *Corvus corax* at early Californian wind farms has contributed to the suggestion that hunting raptors may become 'fixed' on prey so that they may not see turbine blades and hence collide (e.g. Orloff & Flannery 1992). Experiments on visual acuity of raptors and painting of turbine blades (e.g. McIsaac 2001) suggest that some painted patterns may increase conspicuousness, but whether the principle is worthwhile practically is debateable (Young *et al.* 2003a). Moreover, several European studies imply that scavengers (e.g. the Griffon Vulture *Gyps rueppellii*) may be susceptible to collision in some situations (Langston & Pullan 2003, Barrios & Rodriguez 2004) so a predatory lifestyle may not be too influential. Preliminary studies at Altamont implicated casualties being a result of attraction to lattice towers as perch sites (Orloff & Flannery 1992) but detailed analyses with more data (Smallwood & Thelander 2004) have dismissed its importance.

Topography and weather are increasingly highlighted as being influential in raptor collision risk (Johnson *et al.* 2000a, Hunt 2002, Langston & Pullan 2003, Young *et al.* 2003b, Barrios & Rodriguez 2004, Smallwood & Thelander 2004, Hoover & Morrison 2005). Both factors have long been known as important for the movements of migrating raptors (e.g. Smith *et al.* 1985) although realization that they may also be relevant for the movement of resident or breeding birds (and therefore home range use) has only been relatively recent and has been rarely studied (e.g. Jiménez & Jaksic 1993, Bögel & Eberhardt 1997, McLeod *et al.* 2002a,b). More research on topography and weather and their interaction affecting space use and local movements of raptors is clearly desirable: allowing safer micrositing of turbines would be an obvious practical benefit (Johnson *et al.* 2000a, Young *et al.* 2003b).

PREDICTION MODELS

Spatial models

The location of a wind farm is one of the few certainties known to affect the impact of a wind energy scheme on birds (NWCC 2000, Langston & Pullan 2003, Percival 2005). Therefore spatial models that attempt to predict areas of greatest sensitivity for birds at the landscape scale (e.g. Williams *et al.* 1996) can be useful design tools, enabling developments to be located so as to minimize the potential effects on identified key species. At a finer scale, spatial models can be used to evaluate the relative effects of individual turbines within particular locations.

Reliable predictive models can be developed based on data from studies elsewhere. For example, McGrady *et al.* (1997) developed a simple model to predict the ranging behaviour of breeding Golden Eagles based on radio-telemetry studies in western Scotland. This model has been advanced through incorporation of additional parameters, principally

[1]For clarification we follow Band et al. (2005) in defining a bird's avoidance of the whole or part of an operational wind farm as 'displacement' and it's avoidance of a moving rotor when on a path of potential collision as 'avoidance'.

topography, by McLeod *et al.* (2002a,2002b) – the so-called PAT (Predicting *Aquila* Territory) model. The model has the ability not only to predict potentially problematic turbine locations but also the likely overlap with a given wind farm layout and the extent of eagle range use (and therefore the likely extent of range loss should displacement occur). Additionally, it can take account of existing constraints on eagle ranging, for example due to forestry, and hence by comparing predictions of the existing patterns of ranging behaviour with those in the absence of habitat constraints it is possible to gain a useful insight to the scale and distribution of habitat management required in order to mitigate potential wind farm displacement effects. A similar model for Hen Harriers (Madders 2003) is currently being evaluated.

Apart from assisting with site location, design and mitigation, generic models are a good starting point on which to base impact assessments and should be used more frequently as they are cost-effective and can identify potentially problematic locations early. However their accuracy is improved considerably by the inclusion of site-specific information on bird activity patterns. Moreover, there is insufficient knowledge for most species to develop predictive models, and therefore site-specific observations are essential.

Collision risk models

Raptor activity may be a predictor of raptor collision risk when making inter or intrasite comparisons (Erickson *et al.* 2001). If raptor use of a site, or an index of exposure to collision risk is related to a measure of fatality rate, gathered at existing wind farms where comparable data has also been collected, a crude predictive method would be to use the data from operational farms to estimate potential collision fatality, from exposure indices at proposed farms. An illustration of this approach, combining data from Golden Eagles and the Red-tailed Hawk *Buteo jamaicensis* at selected US wind farms, is shown in Fig. 1. Aside of the dubious practice of combining data from different species[2], the main difficulty with this method is that correlation between use and fatality is low at several newer generation wind farms (Erickson *et al.* 2001), and so any relationship may also be heavily influenced by one or two sites where fatalities are high. For example, the high correlation in the example of Fig. 1 ($R^2 = 0.96$) is lost when the two extreme fatality rates and exposure indices outliers (both referring to Altamont) are removed ($R^2 = 0.12$). The absence of useful correlations and potential sensitivity to results from a few sites where fatality and risk exposure may be relatively high, has meant that predictions of collision fatality (which adhere to the basic premise of this approach) have tended to be crude, applying to proposed sites the average or range of fatality rates observed at other sites (e.g. Erickson *et al.* 2003).

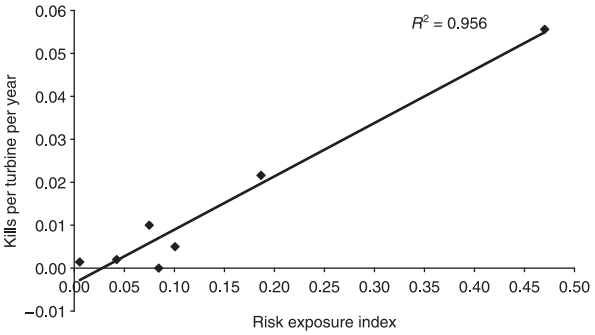


Figure 1. The relationship between an index of collision risk and collision rate as observed at several USA wind farms for Golden Eagles and Red-tailed Hawks using data from Erickson *et al.* (2001, 2002). The index of collision risk is given by mean bird use per 20 min of circular 800 m radius observation plots located within the wind farm site multiplied by the proportion of bird observations at rotor swept height. Collision rate was estimated by searches for collision victims calibrated to account for underestimation due to several biases such as scavenging. Data were used from Altamont, Foote Creek Rim, San Gorgonio Pass, Tehachapi Pass and Buffalo Ridge (see Erickson *et al.* (2001, 2002) for source studies) and to retain clarity for their illustrative purpose have not been transformed for normalization.

It is difficult to question the crudeness of this predictive approach, as it reflects the current uncertainty in the causes of variation in fatalities. However, when a range of potential collision fatality estimates includes a high value, then the scope for disagreement between different stakeholders potentially increases. Cumulative fatality estimates based on such predictions will also exaggerate the range of predicted estimates in the absence of monitoring studies to give 'true' fatality rates; a relevant consideration given the rapid pace of wind energy development in some areas. A desire both for more 'precise' fatality estimates and the ability to contrast

[2]The form of any functional relationship is liable to vary between species, and limited evidence suggests that in, for example, American Kestrel the kill rate may rise more rapidly with increasing risk exposure according to data in Erickson *et al.* (2001, 2002).

the risk posed by different turbine types or other scenarios has lead to the development of predictive models of collision risk (whilst models do not necessarily improve fatality estimate 'precision', they do make the influence of different parameters on collision more explicit: Band *et al.* 2005, Chamberlain *et al.* 2005, 2006).

Tucker (1996a,1996b) produced the first such model, but it has not been widely applied. This may be because the model did not account for the extent of the volume swept by rotor blades, which is clearly pertinent (e.g. Smallwood & Thelander 2004) with the assumption that birds did not avoid collision, which is not the case (e.g. Gill *et al.* 1996). A more recent model (Podolsky 2003, 2005) appears superficially similar, but is more complex than that of Tucker's and incorporate rotor swept volume and avoidance/ 'attraction' factors (although published details are incomplete). This model may be gaining some recognition for practical utility (Schwartz 2004, Smallwood & Neher 2004, Podolsky 2005). A further collision prediction model has been developed by Biosis Research and has been applied in several wind farm assessments in Australia (Organ *et al.* 2002, Timewell & Meredith 2002, Biosis Research 2003). We have not been able to access full details of this model; nevertheless, it appears similar to others but may be more simplistic in some respects (e.g. it does not explicitly acknowledge that not all birds flying through the spinning rotor blades will necessarily be hit).

Initially developed at around the same time as Tucker's model, the 'Band' collision risk model (Band *et al.* 2005) has been used in Environmental Impact Assessments for several years in Scotland, and more recently in the rest of the UK (Chamberlain *et al.* 2005, 2006). An appraisal of the Band model (Chamberlain *et al.* 2005, 2006) has noted that whilst it appears generally robust there is a strong influence of avoidance rates on estimated collision risk and that information on avoidance rates is scant, confirming the conclusions of Band *et al.* (2005). The particular sensitivity of fatality estimates to avoidance is a common issue to collision risk models which incorporate this factor (Biosis Research 2003, Podolsky 2005). Although this problem has to be acknowledged, we do not agree with the notion that it is severe enough to warrant abandoning the use of collision risk models (Chamberlain *et al.* 2005, 2006) because the alternative methods of estimating risk are no more robust (being subject to the same sources of potential error, but without the explicit quantification of relevant parameters) and at least there appears to be a broad consensus that avoidance rates are generally very high (e.g. Biosis Research 2003, Band *et al.* 2005, Podolsky 2005) and that there is scope for their estimation.

With currently available data, one method for estimating avoidance rate is the difference between predicted fatality and observed fatality (avoidance rate = 1 – [observed deaths/predicted deaths assuming no avoidance]). (Note that this means, however, that an avoidance rate calculated using one model may not be transferable to another model.) Bias associated with measuring observed deaths through carcass searches is increasingly highlighted in recent reviews and was rarely accounted for in early fatality studies (Gill *et al.* 1996, Morrison 2002, Langston & Pullan 2003, Band *et al.* 2005, Chamberlain *et al.* 2005, 2006, Whitfield & Coupar 2005) prompting concern that it may affect measures of avoidance rates (Band *et al.* 2005, Chamberlain *et al.* 2005, 2006). Recent studies however, especially in the US, have addressed this problem (e.g. Erickson 2003a, Smallwood & Thelander 2004) although it may be more intractable for offshore sites when remote sensing of avoidance and strike rates may be essential (Desholm 2003). For diurnal raptors, the scope for reliable estimation of avoidance rates using searches for carcasses and other remains may be better than for other bird groups because of their size (rendering carcass detection more likely) and because estimations of utilization rates (relevant to the 'predicted deaths' component of the calculation of avoidance rates) should not be biased by a lack of nocturnal observations, which typifies the vast majority of wind farm studies.

Modelling collision risk under the Band model is a two-stage process (Band *et al.* 2005). Stage 1 estimates the number of birds that fly through the rotor swept disc. Stage 2 predicts the proportion of these birds that will be hit by a rotor blade. Both stages are prone to bias due to the inclusion of relatively simplistic assumptions about bird behaviour. As we noted above, much attention has focused on the adjustment of the output from Stage 2 to allow for a level of turbine avoidance by birds. However, rather less attention has been given to several potentially influential biases associated with Stage 1, although both Band *et al.* (2005) and Chamberlain *et al.* (2005, 2006) discuss some of the possible biases in recording bird activity levels and the sensitivity of collision risk models to bird activity levels, respectively. These potential biases are also obviously relevant to the estimation of avoidance rates

(under the 'predicted deaths' component of the calculation noted above) and other methods which attempt to describe bird utilization rates at wind farms, and are dealt with more fully below.

RECORDING BIRD ACTIVITY

Stage 1 of the Band model is based on field observations that aim to quantify the level of flight activity within the proposed wind farm and immediate environs (typically, the turbine envelope plus 500 m). Under the protocol suggested by Band *et al.* (2005), this is done using time budget data gathered from vantage points strategically located with respect to the area of interest. The method used depends on the type of flight behaviour exhibited.

In Type I models birds are assumed to move along fairly predictable corridors. Examples include diurnal movements of wildfowl, feeding trips by breeding divers (*Gavia* spp.), and spring and autumn migratory bird movements. In Type II models bird movements are assumed to be less predictable, for example ranging activity by territorial and non-territorial raptors.

The collection of data for most Type I models is relatively straightforward, since observations are generally focused on a nest location (e.g. divers) or expected flight corridor (e.g. diurnal movements of geese). An exception is the case of 'broad-front' migratory birds where several vantage points will probably be necessary, aligned perpendicular to the expected line of flight.

In Type II models a critical data collection consideration is to ensure that there is appropriate spatio-temporal coverage of the proposed wind farm area. Thus, vantage points are selected that maximize the visibility of the survey area, ideally such that all parts of the survey area lie within 2 km of a vantage point. In order to minimize observer effects on bird behaviour this is achieved using the least number of vantage points possible. Where feasible, vantage points are distributed outside the survey area; again, this is to minimize any effects of the observer on bird use of the survey area. It is worth noting that this is in stark contrast to the typical survey method used in most US and Australian wind farm studies where an observer records bird activity per unit time within circular plots of variable radius (often 800 m) located *within* the study area (e.g. Johnson *et al.* 2000a, 2000b, Erickson *et al.* 2003): more detailed descriptions of the method can also be found in Reynolds *et al.* (1980) and Bibby *et al.* (1992). Locating observation points within a wind farm study area incurs the possibility of observer effects on measures of bird utilization rates, and was first raised by Orloff & Flannery (1992), but this potential bias does not appear to have been considered subsequently in relevant wind farm studies.

In practice, locating vantage points outside the study area is often incompatible with the requirement for all parts to be within 2 km of at least one vantage point, especially at large sites. This problem is unlikely to be serious, unless data are being collected from a vantage point within the area under observation at that time. It should be noted that overlap in visibility between the various vantage points is inevitable, and whilst this may increase the effective temporal coverage, it may also violate statistical assumptions that the visible areas from each vantage point are effectively independent (e.g. mean values of flight activity from each vantage point are used as input data in the Type II example calculations given by Band *et al.* 2005). Provided that observations are not collected from different vantage points simultaneously, however, this violation should not be serious. The 'circular plot' (or point count: Bibby *et al.* 1992) method is less likely to incur this violation, if observation points within the study area are located randomly (Reynolds *et al.* 1980, Anderson *et al.* 1999) because spatial coverage is not complete, relying on selected observation points to be representative of the study area as a whole. This does, of course, render the circular plot method vulnerable to bias if the selected observation points are not representative through, for example, too few being chosen.

Adequate temporal coverage is achieved through a study design appropriate to the species known/suspected to be present and the phenology and expected levels of flight activity (Anderson *et al.* 1999). It is important that observations are stratified across periods corresponding to different seasons (breeding, migratory, other non-breeding) and, within the breeding period, from different behavioural epochs (for example prebreeding territorial activity, incubation, fledging, post-breeding dispersion). This allows flight activity across the year to be determined with greater precision. In addition, observations should be stratified diurnally across the period during which the target species are known to be active in order that observations are efficient (Anderson *et al.* 1999). This will vary according to species, making it difficult to combine observations for raptors with those for some species of wader, for example. The overall amount of time devoted to vantage point observations should ideally be determined

by undertaking a power analysis of data collected over a reconnaissance period. Further work on this aspect is urgently required in order that adequate sample sizes of data are gathered without unnecessary expenditure of effort.

Additional biases in recording bird activity

Missed observations

Unlike data collection for Type I models, which is usually geared towards a single or related group of species, in Type II models, observers often attempt to gather information on a range of disparate species. If too many target species are chosen or the number of individuals is considerable, there is a danger of observers becoming overwhelmed by a large number of simultaneous flying bouts if the observation protocol is designed to collect time budget data (duration and location of flight, flight height). Similarly, if the species involved have very different behaviours, it may be difficult to adequately scan the landscape for potential activity (for example, searching the skyline for soaring raptors whilst looking for local movements by upland waders). Crucially, it is generally possible to track only one bird at a time. This problem is inherent to many observation methods used in wind farm studies and, importantly, the disparity between recorded and actual activity will be non-linear, becoming greater as actual activity increases. Therefore, bird activity will be increasingly underestimated. This is especially pertinent when many reviews recommend that wind farms should not be located on sites with high bird activity (e.g. Langston & Pullan 2003) and also means, incidentally, that avoidance rates calculated at operational facilities may be underestimated (*cf* Chamberlain *et al*. 2005, 2006).

There are several potential solutions that are not mutually exclusive and their efficacy will vary according to the potential severity of the problem. If the problem is considered not severe, so that temporal overlap in bird activity is negligible, time spent watching target species may be deducted from the overall observation time before activity rates are estimated. Observer number can be increased and labour divided (e.g. Smallwood & Neher 2004). Instantaneous records (Altmann 1974) of the number of birds present can be made at set intervals if bird numbers are high (if numbers are low this method will probably generate a large number of zero values which can be statistically awkward), with more detailed time budget observations collected separately, or in intervening periods on a sample basis (e.g. Johnson *et al*. 2000a).

Observer acuity

A long-standing plea in reviews and methodological examinations of wind farm/bird interaction studies has been for standardization of approach (e.g. Gauthreaux 1995, Anderson *et al*. 1999, NWCC 2000). Inter-observer variation in acuity skill countermands standardization (Gauthreaux 1995) and although flight activity recording protocols can be made consistent to a degree through training, ultimately the method is dependent on a high level of observer field skill. The detection of some species requires experience. In many upland areas in the UK, an endemic problem is that long periods of time can elapse between periods of (sometimes frenetic) flight activity. As a result, interobserver differences may be a large source of bias when comparing studies.

Unfortunately, whilst substantial improvements in avian studies at wind farms have been made in recent years, the issue of observer variability has not been addressed, despite its influence being apparent from the earliest methodological review (Gauthreaux 1995). Hence, this aspect is rarely acknowledged in environmental impact assessments, and even more rarely accounted for. Consequently, while other sources of bias have been recognized in syntheses or 'meta-analyses' of wind farm impacts, observer variability has been ignored (Erickson *et al*. 2002, Erickson 2003b), most liekly because there was little that the investigators could do about it. The problem remains, nevertheless, and has become more acute in recent times as skilled observers become relatively less available as a result of the accelerating pace of wind energy development. We would urge therefore that greater attention is paid to this issue through, for example, increased use of observer training and calibration of acuity skills, with such practices being incorporated in to assessments, which place greater emphasis on documentation of and accounting for observer experience.

Visible area

Under the Stage 1 observation protocol advocated by Band *et al*. (2005) measurements of the airspace within which observations are made can be achieved in a variety of ways and the various approaches can yield widely differing values of flight activity per unit area. For example, some assessments estimate the area of ground visible by eye in the field, whilst others use view sheds generated by GIS software:

the latter is most likely more effective (Band *et al.* 2005). When assessing potential collision risk in areas with varied terrain, view sheds usually describe the visible area of an imaginary laminate suspended at the lowermost height passed through by the proposed turbine rotors (typically 30 m), in order that the potential future risk zone is adequately covered. Interestingly, under the circular plot method as applied in wind farm studies, the characteristic assumption that all airspace is visible seems implicit (e.g. Erickson *et al.* 2002). Given the terrain in which many wind farms are proposed, we doubt if this assumption is always valid even though in some studies plots may be relocated to ensure that at least the majority of the outer limit of the plot can be seen (e.g. Johnson *et al.* 2000a).

Observers generally scan for birds within an arc of around 180° under the Band *et al.* (2005) protocol and thus the actual area observed is often much smaller than that potentially visible. The error this potentially generates is minor compared to that generated by the circular plot method because at any one time, the majority of the presumed observation area will not actually be under observation by a single observer. Surprisingly, this has attracted little critical attention (Reynolds *et al.* 1980, Bibby *et al.* 1992) and while indices of activity may be little affected for comparative purposes, absolute indices of activity are bound to be more seriously affected (lowering estimates of avoidance rate, for example).

Detection of flying birds

An important consideration in generic flight activity observations (as opposed to those focused on particular individuals or species) is that the probability of detection is likely to vary between species, and within species can probably vary according to sex, behaviour, terrain and habitat. Thus detection functions are probably both species and site specific. For example, large, slower flying birds that soar conspicuously above the skyline (e.g. Golden Eagle) are likely to be detected at far greater distances from the observer than small, fast flying birds that are cryptic either due to their plumage or behaviour (e.g. Merlin *Falco columbarius*) (Fig. 2). During observation sessions at vantage points conducted at several proposed wind farm sites in Scotland, for Merlins the area around each vantage point within which birds could be reliably detected appeared to be around 176 ha (within a radius of approximately 750 m: Fig. 2). The observations also suggested that there may be a depressed effect on detection of

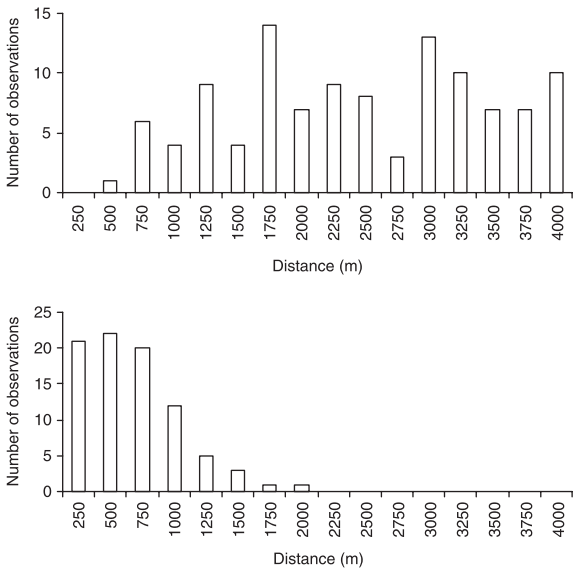


Figure 2. Distance of Golden Eagles (upper chart: $n = 112$) and Merlins (lower chart: $n = 85$) from an observer at point of first detection collected during vantage point observations at several proposed wind farm sites in Scotland. The frequency distribution of observations with respect to detection distance was significantly different between species (Kolmogorov-Smirnov $Z = 6.21$, $P < 0.001$).

Golden Eagles to a distance of at least 750 m from the observer, presumably due to disturbance (Fig. 2). A further problem arises with Merlin because the disturbance effects due to the observer are likely to be disproportionately large within the area where the species is reliably detected. Therefore the problem of reduced detectability can not be overcome simply by using more, closely spaced, vantage points. In practice, vantage point observations may be unsuitable for the purpose of generating realistic estimates of flight activity by such birds; alternatively, incorporating a distance sampling approach (Reynolds *et al.* 1980, Buckland *et al.* 1993) may be necessary.

Similar considerations apply to variation in terrain and habitat, with birds visible over much larger areas in gently undulating open ground than in areas with rugged relief or convex slopes, or where lines of sight are constrained by woodland. This can lead to variation in the detection of species within a single site, although to some extent it may be possible to overcome this through the judicious location of vantage points.

Flight speed

Flight speed is a common input parameter to collision risk models, and whilst it may have only limited influence on fatality estimates and then mainly at

low speeds (Podolsky 2003, Chamberlain *et al.* 2005, 2006) it is typically derived from the literature and based on a relatively small set of data from studies undertaken in situations which may not be relevant to those in which the birds are potentially at risk, for example due to differences in habitat or behaviour (Spaar 1997, Bruderer & Boldt 2001). It is further assumed that birds fly at a constant speed for any given predictive run of a model (e.g. Band *et al.* 2005). Constant speed is highly unlikely since a bird's flight speed varies according to its mode of behaviour (i.e. whether hunting, transiting between locations, displaying, interacting with other birds, etc.), flight height and also due to wind speed (for example, Hen Harriers often use headwinds to quarter the ground methodically for prey at slow flight speeds, then fly downwind at rapid speed prior to turning and resuming a slow quartering flight mode). Collision risk is not linearly related to flight speed (Podolsky 2003, Chamberlain *et al.* 2005, 2006).

Air speed is the actual speed of a bird through the air mass before accounting for the effect of wind or other weather conditions. Ground speed is the speed of a bird compared to the ground after factoring in the effect of wind or other weather conditions and in the context of collision risk is the more relevant metric. Variable ground speed due to wind speed and direction, such as in the Hen Harrier example above, can be accounted for under Stage 2 of the Band model which estimates the probability of a bird passing through rotor blades being struck. Stage 2 has upwind and downwind components (Band *et al.* 2005), and by separately calculating the effect of wind speed decreasing or increasing air speed on upwind or downwind flights, respectively, an average value of the collision probability can be deduced. A further improvement would be to weight the influence of downwind and upwind flights according to their occurrence in the field, but appropriately targeted field observations would be required. Hence, we would make a further recommendation that such observations are collected in wind farm studies, and would encourage more measurements of flight speeds in an appropriate context.

Flight height

The frequency of flights at rotor blade height is clearly relevant to most studies of collision risk at wind farms, whether for computing estimates of activity within a 'risk volume' for a collision risk model (Organ *et al.* 2002, Podolsky 2003, Band *et al.* 2005) or for computing indices of risk exposure (e.g. Erickson *et al.* 2002). Activity at rotor blade height is usually estimated by eye at the point of detection and can then be recorded subsequently at regular intervals. Observers must judge the height of a variety of species over differing types of terrain and habitat at distances ranging from a few hundred metres to several kilometres. There are generally few, if any, landscape features against which the accuracy of height estimates can be checked and parallax effects can further confound the determination of the correspondence between a bird and the ground over which it is flying. Thus, for example, it is possible for an observer to erroneously believe that the bird is flying a short height above a summit ridge when in fact the bird is flying at much greater height over a slope some distance in front of, or behind, the ridge.

Such considerations are consequently important not just for ensuring any error in flight height estimates are addressed by estimates of fatality risk, but also in documenting error in recording the flight paths or location of flying birds because topography may be important in the flight behaviour of some raptors and turbine relocation away from intensively used areas is a potential mitigation tool (Johnson *et al.* 2000a, Mcleod *et al.* 2002b, Young *et al.* 2003b, Smallwood & Neher 2004, Hoover & Morrison 2005). Observers involved in wind farm studies would therefore be usefully trained in flight height and distance estimation (range finder binoculars may not work in all field conditions) and any residual recording error, documented in staged exercises with known-height and known-location observation targets, accounted for in assessment or monitoring studies.

Extrapolation from observed data

There are few published studies describing the activity budgets of upland bird species and potentially influential factors (e.g. Collopy & Edwards 1989). Thus the number of hours per day that birds are potentially active, and the influence of factors such as weather, time of year, breeding status, etc. are poorly understood and so such factors are poorly incorporated in estimates of activity levels. Observations of flight activity are typically collected across the diurnal period and in a range of weather conditions. However, in order to maximize survey efficiency most observations have to be gathered in conditions of good visibility (e.g. Smallwood & Neher 2004) and therefore are unlikely to be representative. The flight activity levels of diurnal raptors

are expected to be lower during poor visibility than at other times, although some species may exploit specific circumstances (for example, misty conditions may allow some raptors to closely approach their prey without being detected). It is likely that many raptors are least active during precipitation (e.g. Watson 1997).

Sample size is an important consideration in designing representative sample studies (Anderson *et al.* 1999). In the case of raptors, wind strength and direction are probably important influences on the distribution of foraging effort, given the reliance of several species on wind conditions for energy-efficient flight (Bögel & Eberhardt 1997, Watson 1997, McLeod *et al.* 2002b, Smallwood & Neher 2004, Hoover & Morrison 2005). It is important therefore that sampling effort is sufficient to document the range of wind conditions which prevail at a site to reduce risk of bias due to stochastic events.

From the above, it is clear that any adjustment made for weather/visibility must be judged on a species by species basis. Because of the difficulty in recording bird behaviour in poor visibility these adjustments are often made in the absence of quantitative information, yet they have a potentially large effect on the predicted flight activity value. Moreover, visibility is likely to strongly influence flying height (Richardson 2000) and this has implications for the proportion of extrapolated flights assumed to be at rotor height. Although it could be argued that raptors hunting in poor visibility would tend to fly below rotor height, their reactions when approaching rotors in such conditions are unknown. These considerations serve to reiterate the suggestion we made earlier in that more research on the factors affecting raptor flight behaviour and movements is sorely needed.

Finally, it must be remembered that although some birds may be less active in poor visibility, the ability of those that continue to fly to detect and avoid operating turbines is likely to be much reduced. In other words, low visibility may have contradictory effects in Stages 1 and 2 of the Band collision model. This is because in Stage 1 (estimating the number of birds that may pass through the rotor blades) without any correction for low bird activity in poor visibility when observations would not be taken, the number of birds at risk will be overestimated, whilst in Stage 2 the estimated probability that a bird passing through the rotor blades will be struck will be underestimated if low visibility decreases avoidance.

CONCLUSIONS

Despite limited evidence that upland raptors are displaced from wind farms in some situations it is apparent that many are at risk of collision with turbine rotors and any overhead lines. The appropriate use of spatial models at the earliest planning stage can help reduce this risk. Modelling collision risk can help to determine the approximate level of mortality likely to result from particular developments, enabling the consequences to be explored for local and regional populations. However, the reliability of collision models is limited by difficulties in gathering appropriate field data and by the large number of assumptions necessary during the modelling process, notably the level of collision avoidance. As a result care must be taken not to over interpret the model outputs, which are probably best used to evaluate different wind farm configurations, for example, rather than taken definitively until further improvements and realistic confidence limits are introduced to model parameterization. We recognize, nevertheless, that as alternative methods for estimating collision risk are less transparent or more subjective and at least as vulnerable to the same potential biases, conceptually robust models are useful tools that can probably incorporate future improvements in knowledge of factors influencing collision and, perhaps more crucially, steer research towards aspects where such knowledge may be most profitably gained.

The method of quantifying flight activity from vantage points is an important element in the assessment of wind farm impacts on birds. However it has a number of limitations, mainly related to observer skills and bird detection, and as a result it probably does not work well for all species and all situations. Alternative methods which survey bird activity also have similar and additional drawbacks. In many cases it may be more appropriate to gather reconnaissance data through vantage point watches before switching to alternative techniques, for example focal observations of individual birds or pairs of birds using visual observations, radio-telemetry or remote sensing by radar.

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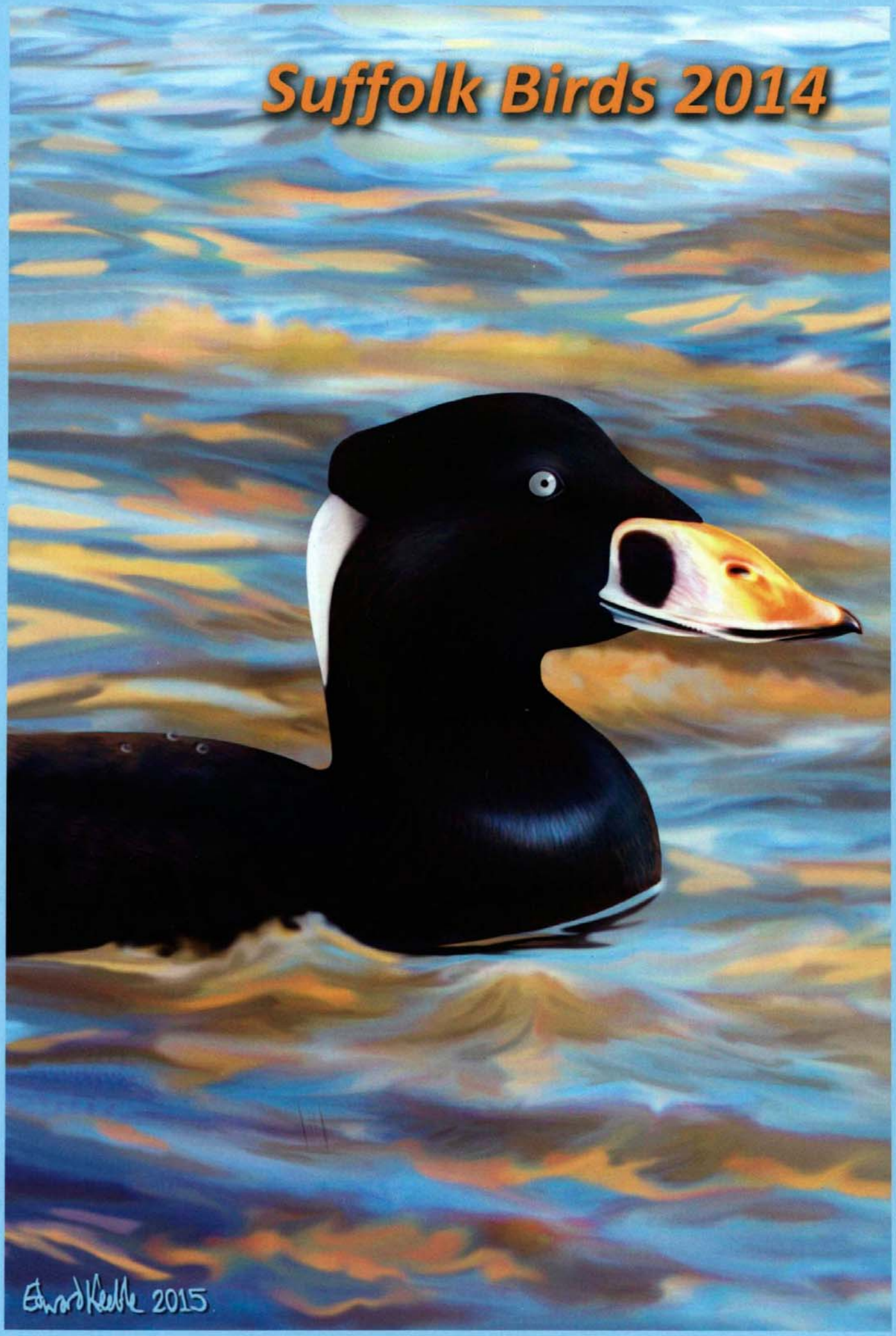
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Suffolk Birds 2014



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A review of birds in Suffolk in 2014

Editor

Nick Mason

Greatly assisted by
Philip Murphy (Systematic List)

Bill Baston (Photos)
Phil Whittaker (Artwork)

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The artwork in this Report is by **Richard Allen**, **Peter Beeson**, **Su Gough, Ed Keeble** and **Brian Small.**

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COMMON QUAIL ***Coturnix coturnix***

Scarce summer visitor and passage migrant. Red List.

Fifteen records, all of singing males were received from eleven sites relating to a probable total of 14 birds. No reports of breeding were received:–

Dunwich: Mount Pleasant Farm, June 3rd (I Salkeld).
Westleton: June 1st (R Drew).
Onehouse: June 22nd (J Walshe).
Pakenham: Puttockshill, July 8th; Aug 5th (S Bishop).
Timworth: July 15th to Aug 2nd (D Cawdron); two, July 19th (S Abbott, N Mason); July 21st (D F Walsh).
Great Barton: Barton Bottom, July 15th (D Cawdron); two, July 18th (N Moran).
West Stow: June 21st (C Gregory).
Lakenheath Fen RSPB: May 29th (West Suffolk Birders).
Icklingham: Berner's Heath, June 28th (T Humpage).
Risby: June 29th (C Gregory).
Higham (near Newmarket): Heath Farm, June 28th (C Gregory).

RED-LEGGED PARTRIDGE ***Alectoris rufa***

Common resident; numbers augmented by releases. Categories C and E.

Some 206 records were received from 65 sites with only five reports of breeding or probable breeding. The highest counts outside of the breeding season, excluding known large-scale releases, were 55 at Falkenham Marshes on October 20th and 58 at Grove Farm, Thurston, on November 24th. The more isolated population on the coast continues to struggle, with records from Landguard being confined to the period of March 13th to 28th when up to three birds were present. On Orfordness there were no sightings reported.

GREY PARTRIDGE ***Perdix perdix***

Formerly common resident, now localised. Red List. Categories A, C and E.

There was a reduction in records received in 2014, down from 108 in the previous year to 82, indicating that this species remains in difficulty. Birds were recorded from 52 sites, one less than the previous year. Confirmation of breeding remains rare with only two reports received; however at least 18 pairs of adults were reported during the breeding season suggesting possible breeders. The highest counts came from West Suffolk, with 34 birds at the BTO Nunnery Reserve on November 13th and 13 at Walsham-le-Willows on December 11th. The sighting of a bird at Landguard on March 9th is the first for the site since 1981.

COMMON PHEASANT ***Phasianus colchicus***

Very common resident; numbers augmented by releases. Categories C and E.

Over 600 records of this common species were received from 110 locations, mainly in the south-east of the county. Breeding however was reported from a mere five sites, although many records relate to sightings within the breeding season. With the widespread releases of birds bred in captivity this remains a very common, although seldom recorded, species.

GOLDEN PHEASANT ***Chrysolophus pictus***

Scarce resident. Categories C and E.

Only one record was received, that of an adult at Euston on January 3rd (E W Patrick). With an absence of records in 2013 it appears as though this species will be extinct in the county in the very near future. There is always the possibility, however, that there are small pockets of woodland where their presence is kept quiet.

RED-THROATED DIVER ***Gavia stellata***

Common winter visitor and passage migrant. Amber list.

The traditional clear-cut temporal and geographic pattern relating to this species' use of Suffolk

waters – in which numbers in the north-east recording area far outstrip those in the area's south-eastern counterpart – was repeated in 2014. It is clearly demonstrated in the following table, which shows peak day-counts for selected months:

	Jan	Feb	Mar	Apr	Sep	Oct	Nov	Dec
North-east	2482	1459	1000	68	16	17	722	690
South-east	279	46	80	6	0	1	50	2

There were 22 records in our north-east area during May, with three in July and two in August.

Bearing this north-east/south-east divide in mind, Orfordness appears to be something of "cut-off" point for this species, with the year's biggest counts usually being amassed to its north and much smaller numbers being encountered to its south, ie closer to the Greater Thames Estuary. However, observers on "the Ness" reported that their seawatching was somewhat "*curtailed*" in 2014 and their "*lower than would be expected*" totals were often exceeded by those to the south, especially at Landguard. In January, the maximum recorded off Orfordness was 25 on 19th; February's maximum was just 12 on 23rd and in March the peak count was 56 north on 15th. The last to be recorded there in the first half of the year occurred on April 20th and the first to return later in the year was seen on October 18th. November's highest count at the site involved 42 north and three south on 23rd, and observers commented that "*remarkably*" there were no records in December.

At Landguard the species was noted up to April 12th and from October 14th to the year's end, with a maximum movement of 232 north and 47 south taking place on January 19th.

BLACK-THROATED DIVER ***Gavia arctica***

Uncommon winter visitor and passage migrant. Amber list.

We may well be seeing a return to this species' true Suffolk status as an "*uncommon winter visitor and passage migrant*" after several years in which numerous reports were submitted, with many believed by the more cynical amongst us to have been erroneous. Some of the county's most seasoned seawatchers have treated such reports with caution – some with outright disdain – and it is fair to say that diver identification is often tricky at best, and sometimes devilishly difficult without extreme care and extremely good views.

In our north-east recording area a total of 24 was reported, a very slight increase over the 20 or so reported in 2013. It is probably wise to remind birders here that fly-by Black-throated Divers are now a description species and that few of these are acceptable records. However, in the south-east recording area the number of reported birds decreased, notwithstanding the fact that what was thought to have been only one individual was recorded several times entering and leaving the River Orwell (Harwich Harbour) off Landguard between January 26th and March 18th (W J Brame *et al.*).

The first autumn passage bird was noted off Thorpeness on September 23rd, with three reports from the north-east area in October, five in November and two in December. All reports concerned "fly-bys" apart from a confiding individual on a small pool in the Westwood Marshes reedbed at Walberswick NNR on December 4th (P Green).

A single at Alton Water present from December 22nd 2013 to January 6th may have been the same bird as that seen on the River Orwell at nearby Freston on January 4th and 6th (N Andrews) and Wherstead on the latter date. In addition to the series of sightings at Landguard referred to above, a report of three there on January 19th raised a few eyebrows. An individual seen at both Bawdsey and Shingle Street on April 4th was the last one to be reported from the south-east recording area in 2014, there being no reports in the area during the second half of the year.

GREAT NORTHERN DIVER ***Gavia immer***

Uncommon winter visitor and passage migrant. Amber list.

This great hulk of a bird again showed its traditional pattern of occurrence, with the bulk of

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A Review of the Population and Conservation Status of British Mammals

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A Review of the Population and Conservation Status of British Mammals

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Executive summary

1. We present the first comprehensive review of the status of British mammal populations for over 20 years. The population size, range size, temporal trends and future prospects of Britain's 58 terrestrial mammals are assessed. Island sub-species and feral mammals are excluded from the main review, but are considered in the appendix.

2. Britain has 44 native species that arrived before the formation of the English Channel. Previously extinct in the wild, the beaver has been reintroduced into Scotland and England in the last decade. The wild boar, also previously extinct, has been the subject of several illegal releases over recent years: the provenance of current populations is unknown. There are 7 species that, although introduced by human activities, are considered naturalised and have formed part of Britain's fauna since at least Roman times. The remaining 7 species are more recent introductions.

3. The geographical ranges of 18 species have increased since 1995; 4 have declined; and 22 have remained stable. A lack of data prevented assessment of the remaining 14 species.

4. Population sizes have increased since 1995 in 15 species; 9 have declined; and 4 have remained stable. A lack of data prevented assessment of the remaining 30 species.

5. All of the species recently introduced to Britain show an increase in geographical range except the brown rat, which is stable, and the American mink, where there are differences between countries. Additionally, all show an increase in population size except the brown rat which appears — on the basis of very poor data — to be stable, and the American mink which appears to be in decline. There are important data deficiencies for all introduced species that need to be addressed urgently.

6. Among native and naturalised species where change could be assessed with reasonable confidence, there have been increases in the geographical range of the following animals:

- Otter, pine marten and polecat.

- Red, fallow and roe deer.
- Greater and lesser horseshoe bat.
- Beaver and wild boar (both of which have become established since the last review, following releases from unknown sources).

Population sizes have increased for the following species:

- Otter, pine marten, polecat and badger.
- Red and roe deer.
- Greater and lesser horseshoe bat.
- Beaver and wild boar.

7. Among native and naturalised species where change could be assessed with reasonable confidence, there have been decreases in the range of the following species:

- Red squirrel.
- Black rat.
- Wildcat.
- Grey long-eared bat.

8. Population sizes have declined for the following species:

- Hedgehog.
- Rabbit.
- Red squirrel.
- Hazel dormouse.
- Orkney vole.
- Water vole.
- Black rat.
- Wildcat.

9. Formally approved Regional Red List assessments, conducted for native species in Great Britain according to the International Union for Conservation of Nature (IUCN) criteria. Approved assessments were also made for the Orkney vole, which is naturalised, because it is officially recognised as an island sub-species; and for the

lesser white-toothed shrew because of uncertainty about whether it is naturalised or native. The assessments placed 26 species in the Least Concern category (meaning that the risk of extinction in the near future is low). Ten native species, plus the Orkney vole, were classified as Threatened (meaning that they face a high risk of extinction). Five native species, plus the lesser white-toothed shrew, were classified as Near Threatened (meaning that they were close to qualifying as Threatened, or are likely to qualify in the near future). Insufficient evidence was available to allow assessment of the other 4 species.

10. All species under review lacked some of the data required for robust estimation of population size. The most common issue was that no information was available on the percentage of potentially suitable habitat within the range that was actually occupied. In these cases, 100% occupancy was assumed, which will usually have led to overestimated population sizes. For example, the Bechstein's bat was assumed to be present in all deciduous woodland, and the red deer in all woodland, within their geographical range.

Robust population density data were lacking for all bats, with the exception of the greater and the lesser horseshoe. There were insufficient data to permit population size estimation at all for the whiskered, Brandt's and Alcathoe bats (cryptic species), barbastelle bat, Leisler's bat, and the potentially migratory Nathusius' pipistrelle bat. One other bat, the noctule, also had a score of zero for population estimate reliability. For this species, estimates could be computed, but they were based on very restricted data, resulting in correspondingly large confidence intervals.

Reliability scores of zero were also assigned to the population estimates for the water shrew, lesser white-toothed shrew, harvest mouse, and weasel, and it was not possible to compute a population estimate at all for the Orkney vole. Overall, 40% of the non-bat species, including all of the shrews, had very poor reliability scores (<=1).

11. Several drivers were associated with temporal changes in population size or range. Fifteen species are currently controlled to reduce their impact on the environment or on other species. Eighteen species have been affected by changes in habitat quality or availability since 1995.

12. The review presents the most up-to-date assessment of population size and status for the 58 terrestrial mammals in Britain. It highlights an urgent requirement for more

research to assess population densities in key habitats, and to assess the percentage of potentially suitable habitat where a given species actually occurs: at present, uncertainty levels are unacceptably high. It is possible that declines in many species are being overlooked because a lack of robust evidence precludes assessment. There is also an urgent need to quantify precisely the scale of declines in species such as the hedgehog, rabbit, water vole and grey long-eared bat, and, where necessary, to identify the causal factors. Finally, effective and evidence-based strategies for mammal conservation and management must be developed.

1 Introduction

The status of terrestrial mammal populations in Britain was last comprehensively reviewed over 20 years ago (Arnold, 1993; Harris et al., 1995). Yet mammals are key components of practically every terrestrial ecosystem in Britain. Small mammals are prey for mammalian predators and avian raptors (Norrdahl and Korpimäki, 1995) and their burrows create nesting sites for bumblebees (Kells and Goulson, 2003). Ungulates shape the landscape through grazing (Palmer et al., 2003), and bats and small mammals can indicate the health of an ecosystem (Pearce and Venier, 2005; Jones et al., 2009). An up-to-date review of the population and conservation status of British mammals, and an assessment of their likely future prospects under changing environmental conditions, is, therefore, long overdue. Whilst there have been, of course, focused studies on particular taxa conducted since the previous review, these do not deliver the broad overview of the relative status of different species necessary for the prioritisation of investment and practical action. This project not only summarises the available evidence derived from hundreds of different research projects, but it also uses — as far as possible — consistent methodologies across species, and is transparent about the approaches and assumptions that have been applied. Using more than 1.5 million individual biological records, this review presents the best available estimates of population size, geographical range, status trends and threats.

Many British mammals are intensively managed. The objective of this management may be to reduce damage to agricultural crops, forestry or other wildlife by species considered to be pests, such as rodents (Labuschagne et al., 2016) and deer (Trenkel et al., 1998), or to prevent the transmission of disease to livestock or humans (Gortázar et al., 2012). Conversely, conservation measures may be required to halt or reverse the decline of a threatened species. These may include protecting species such as bats and the water vole, *Arvicola amphibious,* from the impacts of built development and agricultural change (Roos et al., 2012); preserving the genetic integrity of native species threatened by hybridisation, such as the Wildcat *Felis silvestris* or the red deer *Cervus elaphus*; or helping to secure the long-term future of small populations of pine marten *Martes martes* or red squirrel *Sciurus vulgaris* through translocation. Unfortunately, the evidence base for management is frequently poor. This review highlights species of concern, identifies current and likely future threats, and also explicitly states where conclusions are limited by the lack of sound information.

On a global scale, the beaver is the only British mammal is considered at imminent threat of extinction ('Threatened' under IUCN Red List Criteria (IUCN, 2001)), though the otter *Lutra lutra*, Bechstein's bat *Myotis bechsteinii,* and barbastelle bat *Barbastella barbastellus*, are considered Near Threatened. Perhaps surprisingly, Great Britain is also a stronghold for the non-native Chinese water deer *Hydropotes inermis*, which is increasingly threatened in its native habitat. Despite this apparent lack of threat to the mammals found in Britain in a global context, the state of our wildlife is clearly important ecologically, culturally and morally, and responsibilities towards it are enshrined in national and international law. To inform future planning and conservation action, we have therefore produced a Regional Red List for British mammals using IUCN criteria. This work is presented separately, but the summary information is included within this review. Red Lists are designed to highlight imminent risks of extinction and short-term changes in conservation status ('short-term' being defined as 10 years or 3 generations – whichever is the greater), but they do not account for historical depletion of populations over the longer term. Yet, clearly, long-term change is also of concern, even where population sizes and geographical ranges are currently stable. This review therefore also assesses change since the last comprehensive assessment of geographical range (Arnold, 1993) and population status (Harris et al., 1995) more than 20 years ago.

Accurate information on the distribution and population density of mammal species, as well as insight into the temporal trends, is vital for the development of effective management (Gibbs et al., 1999; Collen et al., 2013). These data are also required to enable the UK to fulfil its international reporting obligations, for example under Article 17 of the Habitats Directive. Even post-Brexit, there will be obligations for monitoring and reporting under domestic law and international treaty (such as the Bern Convention). The amount of detail required depends on the intended scale at which the data will be used (Dickinson et al., 2010). Whilst species may show wide variation locally, a robust estimate of the national status of a species is important for conservation planning.

National surveys tend to be limited to the identification of a species' distribution or, sometimes, where effort is standardised, they identify the percentage of occupied habitat within a species' range. Surveys of this nature have been carried out for several mammalian species, including the otter (Strachan, 2007; Crawford, 2010; Strachan, 2015a), the polecat *Mustela putorius* (Birks and Kitchener, 1999b) and the pine marten (Croose et al., 2013; Croose et al., 2014). These surveys give a useful snapshot of the status of a species' distribution, and they also provide a baseline from which to measure change (Lindenmayer and Likens, 2010). When consecutive surveys are conducted, it is possible to measure

temporal trends in a species' distribution or, where counts are taken, monitor changes in relative population size or occurrence. Schemes such as the National Dormouse Monitoring Programme (NDMP) have provided monitoring over considerable periods (the NDMP has run since 1988), and otters have been the subject of repeated country-wide surveys (e.g. Crawford, 2003; Strachan, 2007; Crawford, 2010; Strachan, 2015b) that have allowed for the identification of temporal changes in distribution. Badgers *Meles meles* are monitored via counts of the number of badger setts (Cresswell et al., 1990; Wilson et al., 1997; Judge et al., 2014), and bat species via a combination of acoustic field survey data, hibernation surveys and roost counts (National Bat Monitoring Programme). These permit inferences to be made about relative changes in population size. However, absolute measures of population size would require additional data on the number of badgers per social group, and density of bat maternity roosts, respectively.

Absolute measures of population size rely on estimates of population density or total counts in a range of habitat types. The obtaining of density information requires a larger investment of time and effort than the determining of presence alone, and often necessitates repeated surveys. The effort needed to determine density is higher still where species are elusive or rare (Zylstra et al., 2010; Hui et al., 2011). These limitations mean that the scarce resources available for monitoring have tended to be directed at measuring changes in distribution or relative population size, rather than estimating absolute population sizes. However, there are purposes for which absolute population sizes rather than indices of relative change are extremely important. For example, determining whether wildlife fatalities (such as from collision with vehicles or wind turbines, or from culling) are likely to have a material impact on local populations depends on having reasonable estimates of population sizes. The prioritisation of conservation and management actions also often requires an understanding of population sizes: two of the three IUCN Red List Criteria, for instance, are based on knowing the number of mature individuals.

Where they occur, assessments of population size or density are often smaller in scale than distribution surveys, taking place at a single site or a small number of locations rather than at a regional, landscape or national scale. Limitations in the resources available, as well as the behaviour of the species under study, often dictate the methods employed. For easily observed animals, surveys can be carried out using direct observation. These surveys may take the form of direct attempts to count the local population (e.g. for deer species; Putman et al., 2011). Alternatively, inferences can be drawn from observing subsets of the population: refinements such as distance sampling — which adjusts for the declining probability of detection with increasing distance from the observer — help to improve the

quality of such estimates (Borchers et al., 2015) (e.g. for mountain hare Lepus timidus; Knipe et al., 2013). The new Mammal Mapper app. for mobile phones, provided free of charge by the Mammal Society, is designed to encourage large-scale citizen science participation in distance sampling surveys and to improve the quality of data available on habitat-specific mammal densities.

For small mammals, measures of population density tend to involve live-trapping. The total number of animals trapped during a single trapping session can provide a proxy for population density in the form of the total of individuals trapped, or 'minimum number alive' (MNA), when divided by an estimate of the spatial extent of the trappable population (the 'effective trapping area'). This area is, however, difficult to define and leads to density estimates which are often considered unreliable (Efford, 2004). Where it is possible to conduct multiple trapping sessions, capture-mark-recapture models can be used to estimate density from the capture history of individuals within the trapped population, although multiple surveys inevitably require more time and effort, and certain assumptions about the population and capture process must be met (see Amstrup et al., 2005). Where species are rare or elusive, indirect methods of detection may be required (Pereira et al., 2010). Camera trapping (e.g. for wildcats; Hetherington and Campbell, 2012; Kilshaw et al., 2015), or counts of non-invasively collected samples such as faeces (Jarman and Caparano, 1997; Gormley et al., 2011) negate the need to observe or disturb the target animals. Aside from the notable exception where animal behaviour conforms to strict model assumptions (Rowcliffe et al., 2008), individual identification is required for absolute measures of population density. Recent advances in molecular techniques have enabled individual identification from non-invasive sources such as hair, faeces and feathers (Waits and Paetkau, 2005), although the costs required to process such samples are still relatively high and processing the samples is time-consuming. In summary, despite substantial effort to detect relative trends in distribution for some species such as otters (e.g. Crawford, 2003; Strachan, 2007; Crawford, 2010; Strachan, 2015b), the investment of time, effort and funding for large-scale assessments of the status and trends for mammalian species means that such assessments are few and far between in Britain.

Battersby (2005) reviewed the monitoring schemes in place under the Tracking Mammals Partnership, as well as those proposed, and reported the available trends in relative abundance and distribution. That review did not, however, provide updated estimates of population size. Reliance has therefore been placed on the assessment of population size, status and trends undertaken in the mid-1990s by Harris et al. (1995). For that assessment, population density estimates for all terrestrial mammals in Britain were taken from peer-

reviewed literature and expert opinion. Population sizes were then estimated by combining habitat-specific density estimates with the area of available habitat within the species' distribution. Habitat data were derived from the land classes devised by the Centre for Ecology and Hydrology (formerly the Institute of Terrestrial Ecology) (Bunce et al., 1981a; Bunce et al., 1981b; Bunce et al., 1996), and distributions were based on data deposited up to March 1992 which were reported in the Mammal Atlas (Arnold, 1993). Information on population trends for different mammal species were quoted where the information was available in the literature, or were derived through expert knowledge; current distribution and the legal status of each species were also described. We have followed broadly the same approach in the current review.

Recently, Croft et al. (2017) have tested an approach based on habitat suitability modelling to produce density estimates for mammals in Great Britain. In a first step, habitat suitability models were built using environmental data coupled with occurrence information derived from the National Biodiversity Network (NBN). These habitat suitability scores were then linked with habitat-specific density estimates to produce an abundance estimate. The current report and Croft et al. (2017) shared the requirement for habitat-specific density data, and therefore suffer the same constraint that for many species the required information is simply not available. This problem is particularly acute for bats, and so, in this review, an alternative approach that did not depend on habitat-specific information was deployed. Both studies also suffer from a lack of information about the level of occupancy in a given habitat across a species' range: for very widespread species such as the field vole *Microtus agrestis*, this may not be a major issue; for more patchily distributed species, such as the red deer, it could introduce important errors.

However, the approach used by Croft et al. (2017) differs from the present review in several important respects:

- We had access to data at a very much greater spatial resolution, whereas the data used to derive the habitat suitability models in Croft et al. (2017) were available only at 1km or 10km resolution.
- We were able to include datasets not available via the NBN.
- Our assessments were based on pre-breeding density data only, to ensure comparability with the Harris review.
- The present review used a rigorous process of data cleaning, based on the input of many recognised experts to generate a smoothed distribution map of distribution.

The habitat suitability models in Croft et al. (2017), in contrast, are based on the data as presented in NBN.

There are strengths and weaknesses to both approaches. The advantage of habitat suitability modelling is that it can allow extrapolation to poorly surveyed areas on the basis of other environmental variables (temperature, habitat, etc.). However, sensible models can only be built where there are sufficient data to parameterise them; and for many British mammals the evidence is lacking. Also, the spatial resolution of publicly available data is a major constraint, particularly where species are unlikely to be uniformly distributed across a grid square because of particular habitat requirements. The figures presented in the two reports are not directly comparable. Nevertheless, it is reassuring that for many species the population size estimates are within the same order of magnitude.

Population status is a dynamic quantity, and many mammals in Britain are thought to be in decline (e.g., the hedgehog *Erinaceus europeaus* and the red squirrel (Roos et al., 2012; Gurnell et al., 2015b)), whereas several others are apparently increasing in population size and range (e.g., the pine marten and the polecat (Croose et al., 2014; Croose, 2016)). Population assessments from Harris et al. (1995) are, therefore, unlikely to reflect the current status of British mammals, but are the most recent reference for most mammals and are still quoted in the current literature. In this review, we provide the following:

- A current distribution map for each species, using presence data from 1995 to 2016, smoothed using an alpha-hull approach.
- An assessment of current conservation status.
- Estimates of habitat-specific population density from the recent literature (1995-2016) or from expert opinion.
- Current estimates of population size for England, Scotland, and Wales, and the total for Great Britain.
- A critique of these estimates, with a review of data deficiencies.
- A review of the temporal trends in population size since the last review in 1995 (which was based on data collected up to 1992), and the drivers leading to the observed trends.
- A review of the future prospects of each species (see Appendix 7).
- Regional Red List status for Great Britain that has been formally approved by the Inter-Agency IUCN Red Listing Group.

The review presents the most up-to-date assessment of population size and status for the 58 terrestrial mammal species resident in Britain. Accounts are provided in Appendix 6 for a further 9 species. These are either island sub-species (the Skomer vole), feral animals (the feral ferret, sheep and goat), vagrant visitors to Britain (the parti-coloured bat and Kuhl's pipistrelle), or are only present as managed populations or occasional individuals (the reindeer, wallaby and raccoon). Further, it identifies key areas for further research where the data required to assess population size accurately are lacking, and highlights the future prospects of each species and urgent requirements for conservation action.

4 ERINACEOMORPHA

4.1 Hedgehog *Erinaceus europaeus*

Habitat preferences

The hedgehog is found in most habitats, although it is increasingly associated with urban areas, and is often observed in gardens and amenity grasslands. Because it is a mobile, generalist species, road fatalities are fairly common, but these have declined over recent decades, providing some indication of a declining population (Wembridge et al., 2016b). Hedgehogs in rural villages have recently been shown to have small home ranges compared with other habitats, presumably because of greater foraging resource and nest site availability (Pettett et al., 2017). Density is higher in areas with amenity grassland compared with pasture (Micol et al., 1994; Young et al., 2006; Parrott et al., 2014); key prey items, including earthworms, ground beetles and tipulid larvae, are important in determining their distribution. The presence, and abundance, of badgers — one of the few natural predators of hedgehogs — is inversely linked with hedgehog distribution patterns (Doncaster, 1994; Young et al., 2006; Parrott et al., 2014; Trewby et al., 2014).

Status

Native.

Conservation Status

- IUCN Red List (GB: VU; England: [VU]; Scotland: [VU]; Wales: [VU]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 4.1a. Gaps in the species' distribution in Scotland are likely to represent areas lacking survey effort, rather than true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

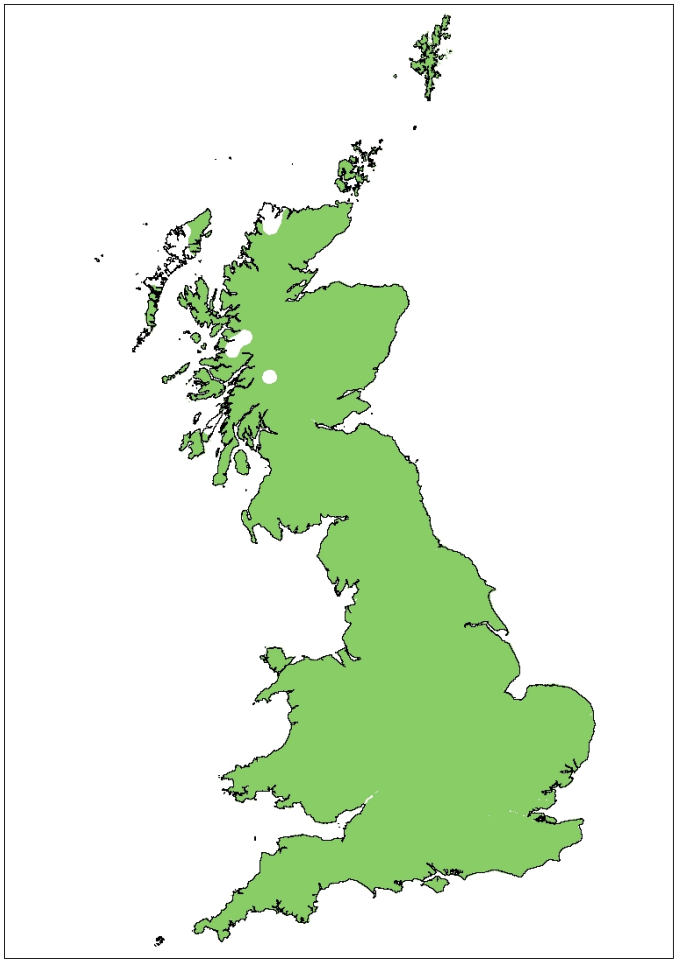


Figure 4.1a Current range of the hedgehog in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Hedgehog occurrence has been monitored by a number of nationwide surveys, where percentage occupancy is the percentage of unique locations where hedgehogs or field signs were observed. Several of these studies have measured hedgehog occurrence in gardens and amenity grassland (Micol et al., 1994; Roos et al., 2012; Parrott et al., 2014). The results of Living with Mammals (PTES), Garden Birdwatch (BTO), Make Your Nature Count (The Royal Society for the Protection of Birds) and HogWatch (PTES) are summarised by Roos et al. (2012). The mean percentage occupancy for the most recent year (per country where available) from these surveys, and from Micol et al. (1994) and Parrott et al. (2014), was used as percentage occupancy for urban and gardens. The HogWatch data came from 2006, as very few responses were received in subsequent years.

Percentage occupancy was also available for some other habitats, including pastoral land, grassland, woodland, arable land, roads and along waterways. However, most of the surveys incorporate a variety of habitats, rather than providing habitat-specific occupancy values. Data from a range of studies conducted since the Harris et al. (1995) review were summarised by Roos et al. (2012). Mean percentage occupancy for the most recent year (per country where available) from these surveys, and from Hof and Bright (2012) and Parrott et al. (2014), was used as percentage occupancy for all other habitats.

Results

Ten papers were returned from the literature search in total. Two papers contained pre-breeding population density estimates, although one paper only contained density estimates for blackland (peatland) and machair habitats that are specific to the north west of Scotland and the offshore islands (Jackson, 2007), and so was not included in the analysis. Three papers contained measures of percentage occupancy, two contained post-breeding density estimates, one contained relative abundance measures, and two presented habitat suitability measures.

Table 4.1a Median density estimates for hedgehogs with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n*	%Occ†
Urban and gardens	13,800	0.54	0.33	1.54	Parrott et al. (2014)	4	49%
Improved grassland	72,800	0.04	<0.01	0.10	Parrott et al. (2014)	4	37%
Arable and horticulture	62,600	5	-	-	Harris et al. (1995)	1	37%
Broadleaved woodland	13,100	40	-	-	Harris et al. (1995)	1	37%
Coniferous woodland	14,400	5	-	-	Harris et al. (1995)	1	37%
Unimproved grassland	12,200	40	-	-	Harris et al. (1995)	1	37%

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 4.1b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates. Values were obtained by multiplying population density estimates in Table 4.1a with the area of habitat within the species' distribution, and adjusting for occupancy. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	123,000	[320,000]	-	-
Scotland	47,100	[145,000]	-	-
Wales	18,800	[56,800]	-	-
Britain	**189,000**	**[522,000]**	**-**	**-**

Critique

Very few recent density estimates were found for hedgehogs: data were available only for improved grasslands and urban areas (Parrott et al., 2014). Median density estimates for both of these habitats are supported by four individual density estimates from one study. Despite improved grassland being the most common habitat across the species' distribution (38%; Figure 4.1b), the population size is largely determined by density estimates in unimproved grassland and broadleaved woodland. These estimates, which are based on Harris et al. (1995), are substantially higher than for other habitats. However, the density estimates presented in Harris et al. (1995) for improved grasslands and urban areas ($5km^{-2}$ and $10km^{-2}$) are an order of magnitude greater than more recent estimates for these habitats, and were largely based on the expert opinion expressed in Burton (1969). It is therefore concluded that the estimates from Parrott et al. (2014) are more reliable. The population densities in Table 4.1a, and subsequent population sizes in Table 4.1b, are therefore likely to be overestimates.

As the current estimate is largely based on density estimates from Harris et al. (1995), the reason for a reduction in population size is likely to be the use of percentage occupancy data, rather than differences in population density, as occupancy data were not used in Harris et al. (1995). Percentage occupancy was taken from a large number of sources (see Micol et al., 1994; Hof and Bright, 2012; Roos et al., 2012; Parrott et al., 2014), although a mean value was used for most habitats. Percentage occupancy ranged from 0% to 81% across all studies, in all parts of Great Britain, so stratification by area, with habitat-specific occupancy data, would provide a significant improvement to the current analysis.

If we assume the estimate provided in Harris et al. (1995) to be the best estimate available for that time period, then applying the decline in relative abundance of hedgehogs estimated by Roos et al. (2012) from citizen-science surveys (40% every 10 years) would result in a total population size of 560,000 in Britain, which is close to our estimate. This extrapolation is, however, subject to uncertainty in the original population size as well as in the trend data. The species' range has remained relatively stable since 1993 (Arnold, 1993), suggesting that declines in population size are owing to reduced density or occupancy. Inferences about density declines are supported by the relative lower density estimates in Parrott et al. (2014) compared with those from studies in continental Europe, which are between $2km^{-2}$ and $300km^{-2}$, depending on habitat type (Huijser and Bergers, 2000). The recent density estimates are also lower than those from Harris et al. (1995), although the latter were largely based on expert opinion rather than empirical data. As well as suggesting that hedgehog

density is highly variable, these figures highlight the need for more empirical data on the population density and occupancy of hedgehogs to improve confidence in the current population density and subsequent size estimates. The reliability of the population density estimates is given in Table 4.1c. To reflect the uncertainty arising from the use of expert opinion in Harris et al. (1995), the density estimates classified as having been derived from a restricted range.

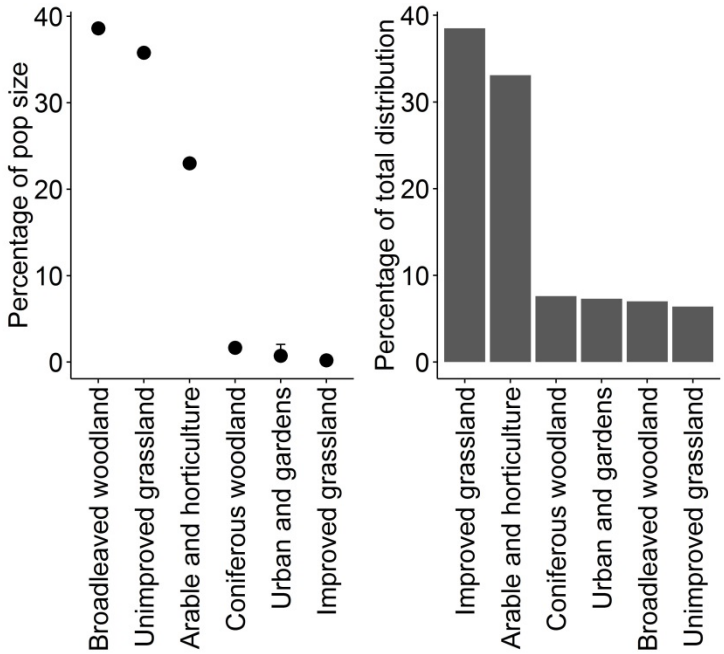


Figure 4.1b Left: The percentage of the total population of hedgehogs derived from each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 4.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat			
			Broadleaved woodland	Unimproved grassland	Improved grassland	Arable and horticulture
Location of study sites	0	Estimates from one location				
	1	Estimates restricted	1	1	1	1
	2	Estimates widespread				
Sample size	0	<10 density estimates	0	0	0	0
	1	10-30 density estimates				
	2	>30 density estimates				
Occupancy data available?	0	No				
	1	Yes	1	1	1	1
		Habitat score	**2**	**2**	**2**	**2**
Overall reliability score			**2**			

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) provided an estimate of 1,555,000 hedgehogs in Britain. The current estimate is substantially lower (-67%). Nationally, there are changes between the two reviews in the estimated availability of key habitats (broadleaved woodland, improved and unimproved grassland, and arable land), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a population size of 423,000, which is a 73% decrease.

Even greater declines would be implied were the estimates based entirely on the density data given by Parrott et al. (Parrott et al., 2014). It is therefore possible that population size has declined further than estimated here, but all current estimates are very uncertain.

Other evidence of changes through time

The relative abundance of hedgehogs has been monitored by several organisations in the UK over the last 25 years. These studies, which used different methodologies and were of varying duration, were reviewed by Roos et al. (2012). There was considerable inter-annual variation within each study, and also variation between them — annual declines ranged from a mean of 1.8% to 10.7% — but there was consistency in the direction of the effect. The authors inferred a decline in relative abundance of 40% in 10 years. However, the scale of this decline contrasts with another study which used non-systematic occupancy records from Biological Records Centres and adjusted for survey effort (Hof and Bright, 2016). Here, a decline of between 5.0% and 7.5% was found for England over a 40- year period, which would mean a maximum decline of 1.9% over 10 years.

Table 4.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995), and trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		All countries*		
	Data deficient				

*Roos et al. (2012).

Drivers of change

Table 4.1e Drivers of population change for hedgehogs between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Agricultural intensification results in a reduction in invertebrate biodiversity and loss of hunting opportunities. There may also be an impact of pesticide use, including in gardens, on prey abundance.	Hof and Bright (2010)	Negative
Habitat availability.	Agricultural intensification results in the loss of hedgerows and field margins, with the effect of reducing habitat connectivity and availability of refugia.	Hof and Bright (2010)	Negative
Vehicle collisions	Road casualties are likely to have had an important effect on local populations.	Huijser and Bergers (2000) Wembridge et al. (2016b)	Negative
Predation/ competition.	Predation and possible competitive exclusion by badgers.	Parrott et al. (2014) Trewby et al. (2014)	Negative

Data deficiencies

Table 4.1f Areas where further research is required to improve the reliability of population size estimates for the hedgehog.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	Arable and horticulture Broadleaved and coniferous woodland, unimproved grassland	Density estimates were taken from Harris et al. (1995).
Density estimates do not represent within-habitat variability.	All habitats	Densities range from 2km^{-2} to 300km^{-2} in continental Europe (Huijser and Bergers, 2000).

Future prospects

Table 4.1g An assessment of the future prospects of the hedgehog, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline*

*Causes of historic declines are poorly understood. There is no evidence that the trend is likely to change.

5 SORICOMORPHA

5.1 European mole *Talpa europaea*

Habitat preferences

The mole is a highly adaptable species, found in most habitats where invertebrate prey is present and the soil is sufficiently deep to allow tunnel construction. In low-lying areas and regions prone to flooding, it can build more permanent ‘fortresses’ which contain a nest chamber above the level of the surrounding land. These are often provisioned with stores of decapitated worms for consumption when the surrounding area is flooded or frozen. Originally an inhabitant of broadleaved woodlands, the mole thrives in pastures and on arable land. It lives at low densities in coniferous forests, on moorland and in sand-dune systems, probably because of the paucity of suitable prey (Harris and Yalden, 2008). Home ranges are small — around 0.2ha for females and 0.3ha for males — and adults rarely disperse once a territory is established (Stone and Gorman, 1985). Although it is aggressive towards intruders, agonistic encounters are very rare (Gorman and Stone, 1990). Earthworms are the most important prey item, particularly in winter, whereas in summer up to 50% of the diet is formed of insects (adult and larvae) (Funmilayo, 1979).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

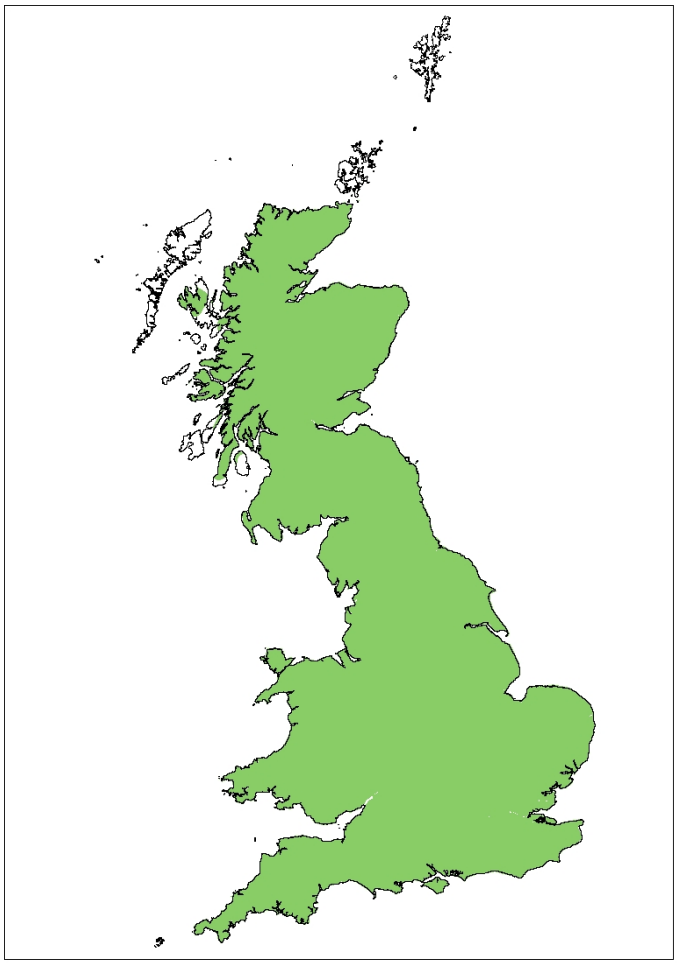


Figure 5.1a Current range of the mole in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Two density estimates from Harris et al. (1995) fell into the LCM2007 'Improved Grassland' category; these were 1.3ha^{-1} for lowland improved grassland, and 4ha^{-1} for semi-improved grassland. For the current analysis, a mean value of 2.65ha^{-1} was used.

Results

No papers were identified with population size estimates for moles, nor were any estimates obtained from expert opinion. The population density estimates (Table 5.1a) are therefore taken from Harris et al. (1995). These were based on expert opinion, where each habitat

was deemed ‘poor’ or ‘good’, and assigned a density of 1.3ha^{-1} or 4ha^{-1}, respectively. Population size estimates are provided in Table 5.1b.

Table 5.1a Median density estimates for moles, with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Arable and horticulture	6,250,000	1.30	-	-	Harris et al. (1995)	1	n/a
Broadleaved woodland	1,310,000	4.0	-	-	Harris et al. (1995)	1	n/a
Coniferous woodland	1,430,000	1.30	-	-	Harris et al. (1995)	1	n/a
Dwarf shrub heath	1,820,000	1.30	-	-	Harris et al. (1995)	1	n/a
Unimproved grassland	7,250,000	4.0	-	-	Harris et al. (1995)	1	n/a
Improved grassland	1,150,000	2.65	-	-	Harris et al. (1995)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 5.1b Area of suitable habitat (not adjusted for occupancy) within the species’ range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 5.1a with the area of habitat within the species’ distribution. It was not possible to calculate confidence limits as a measure of variance was not available from Harris et al (1995).

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	11,500,000	[24,300,000]	-	-
Scotland	5,800,000	[12,200,000]	-	-
Wales	1,910,000	[4,900,000]	-	-
Britain	**19,210,000**	**[41,400,000]**	**-**	**-**

Critique

No percentage occupancy data were available; therefore, the population size is overestimated for this species. 46% of the estimated population size for moles was derived from improved grassland habitat, with a further 19% from arable and horticulture. These habitats represent 38% and 33% of the species' range, respectively (Figure 5.1b). As the density estimates for these habitats were derived from Harris et al. (1995), a sensitivity analysis was not possible. To assess reliability, we have considered the population density estimates from Harris et al. (1995) to be the expert opinion of the authors and, therefore, to represent a restricted area of the species' range (Table 5.1c).

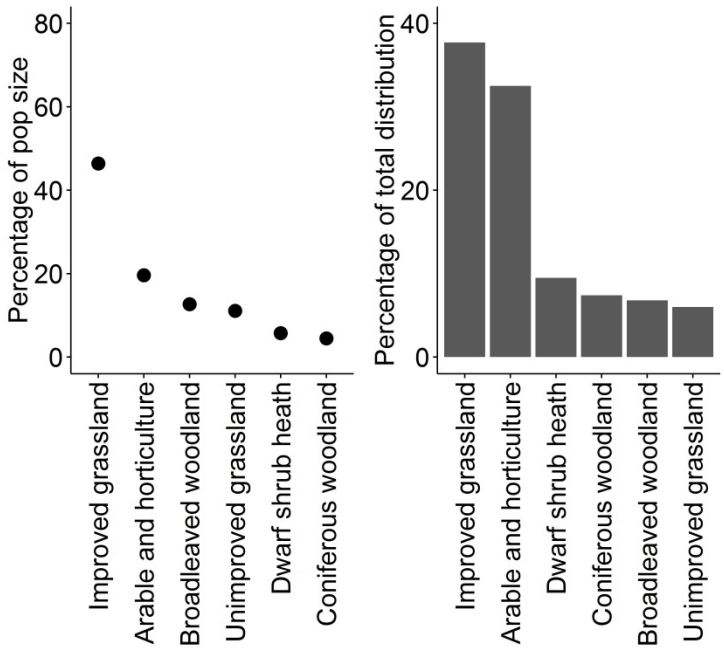


Figure 5.1b Left: The percentage of the total population of moles derived from each habitat type. It was not possible to compute error bars for this species. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 5.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Improved grassland	Arable and horticulture
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	0
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**1**
		Overall reliability score	**1**	

* Populations may be unstable owing to inter-annual cycles, documented fluctuations in population size, or as a result of management.

Changes through time

Comparison to Harris et al. (1995)

Total population size was estimated to be 41,400,000 in Britain, with 24,300,000 in England, 12,200,000 in Scotland and 4,900,000 in Wales. The density estimates used in the current analysis are taken from Harris et al. (1995), so any differences are entirely owing to changes in the species' distribution and land classification. Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a population size of 38,400,000, and a 23% increase in population size since 1995. However, a lack of confidence intervals means that it was not possible to assess whether the difference across time is significant.

Other evidence of changes through time

Mole signs have been recorded as part of the BTO Breeding Bird Survey since 1995. The number of 1km survey squares with signs of moles was 7% in 1995, 32% in 2003, and 18% in 2015. The extent to which differences between survey years reflects variation in recorder effort or true biological variation is not known. Nor is it possible to relate presence of signs to population estimates. A summary of trends in population size and range is provided in Table 5.1d.

Table 5.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries		

Drivers of change

Table 5.1e Drivers of population change for moles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Habitat quality.	Intensification of agricultural practices (ploughing/re-seeding) may reduce food (earthworm) density.	Edwards and Lofty (1972)	Negative
Habitat availability.	The loss of hedgerows, through neglect or removal, and reduction in unimproved grassland may reduce availability of refugia.	Harris et al. (1995)	Negative

Data deficiencies

Table 5.1f Areas where further research is required to improve the reliability of population size estimates for the mole.

Data deficiencies	Habitat	Details
Density estimates more than 10 years old.	All habitats	All population density estimates are taken from Harris et al. (1995).
Density estimates do not reflect within-habitat variability.	All habitats	No ranges or confidence limits were available for the density estimates.
No occupancy data.	All habitats	

Future prospects

Table 5.1g An assessment of the future prospects of the mole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline
Range	Stable
Habitat	Stable/Decline

5.2 Common shrew *Sorex araneus*

Habitat preferences

The common shrew is found in most terrestrial habitats, providing that some low vegetation cover is available. It is most abundant in thick grass, bushy scrub, hedgerows and broadleaved woodland. Fallow land, roadside verges and urban habitats are colonised rapidly. At high altitudes, it is occasionally found among heather and more frequently in stable scree (Harris et al., 1995). Its very high energy requirements, the result of its high

surface area:volume ratio, means that it requires habitats with high invertebrate abundance. It may therefore be negatively affected by changes to agricultural practice and/or pesticide use which reduce its prey availability. There is very little research in the recent literature: most records are reported as part of multi-species small mammal studies.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales; [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 5.2a. Gaps in the species' distribution throughout the mainland are likely to represent areas lacking survey effort, rather than true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

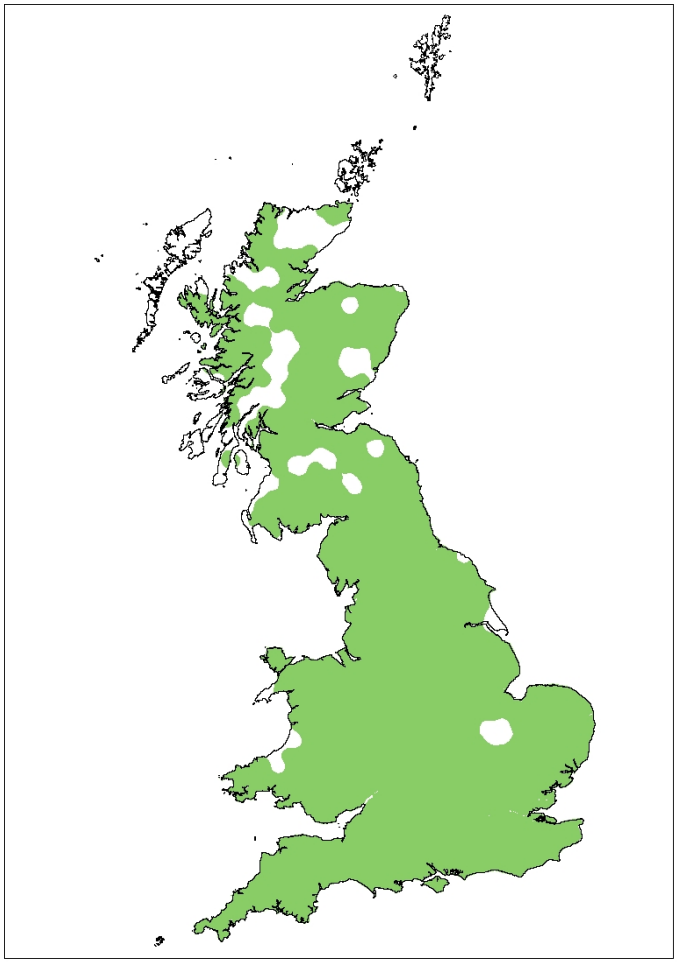


Figure 5.2a Current range of the common shrew in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Specific methods

As many small mammals in arable land are known to use field margins as primary habitats, with cropped areas incorporated into home ranges only before the harvest (Tattersall et al., 2001), we assume that the population within fields will be captured in the estimate for hedgerows. Arable land is therefore excluded from the analysis.

Results

The literature search identified eight relevant papers, four of which contained pre-breeding density estimates, three contained post-breeding estimates of density, one contained percentage occupancy for hedgerows (Gelling et al., 2007), and one containing estimates sourced from papers already included in the current analysis. Population density estimates are provided in Table 5.2a, and population size estimates in Table 5.2b.

Table 5.2a Median density estimates of common shrews with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerow length and density within hedgerows are presented as km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	1,260,000	2.0	0.50	7.25	White and Searle (2007)	5	n/a
Coniferous woodland	1,210,000	1.25	0.25	2.75	White and Searle (2007)	5	n/a
Dwarf shrub heath	1,480,000	0.50	0.25	3.75	White and Searle (2007)	5	n/a
Unimproved grassland	1,040,000	4.80	4.0	9.50	Pernetta (1977)	9	n/a
					Churchfield et al. (1995)	8	
					White and Searle (2007)	10	
Bog	555,000	12.0	0	35.0	Shore and Mackenzie (1993) ††	1	n/a
Urban and gardens	1,350,000	1.95	0.4	3.95	expert opinion	2	n/a
Fen, marsh and swamp	8,000	0.50	0.05	1.0	expert opinion	1	n/a
Improved grassland	6,660,000	0.10	0	0.50	expert opinion	1	n/a
Hedgerows	460,000 (km)	7.70	2.07	13.9	Kotzageorgis and Mason (1997)	9	37.8%
					Flowerdew et al. (2004)	9	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.
†† Median density and confidence limits using data from this paper were provided by the authors.

Table 5.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 5.2a with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Area of suitable habitat (ha) *	Population size	-95%CI	+95%CI
England	7,480,000	[11,000,000]	3,520,000	29,500,000
Scotland	4,340,000	[7,690,000]	1,980,000	22,900,000
Wales	1,740,000	[2,330,000]	1,010,000	6,120,000
Britain	**13,600,000**	**[21,100,000]**	**6,520,000**	**58,500,000**

* The lengths of hedgerows are 394,000km in England, 17,100km in Scotland, and 48,600km in Wales.

Critique

No percentage occupancy data were available for most habitats; therefore, the population size estimate for this species is an overestimate. Although most of the land cover within the species' range is improved grassland (48%; Figure 5.2b), population density in this habitat is thought to be small (0.1ha^{-1}, 95%CI = 0-0.5; Table 5.2a). This density estimate is based on the opinion of one expert; any uncertainty in this estimate will affect the population size estimate. In addition, at least some of the animals included within density estimates for improved grassland are likely to be individuals that also live in hedgerows, potentially introducing an element of double counting. Deriving robust estimates for improved grassland is therefore considered a research priority. Most of the estimated population is derived from unimproved grassland habitat (24%), for which the average density estimate is supported by 18 estimates; and from bog (32%), for which values were based on data from Shore and Mackenzie (1993) provided by the authors. The population density for bog habitat reported in Harris et al. (1995) was markedly lower, at 0.5ha^{-1}, although this value was based on expert opinion rather than empirical data. An increase in population density of this magnitude in the last 20 years is unlikely, and so further data are urgently required to increase confidence in the density estimate for this habitat. Reliability scores for unimproved grassland and bog are provided in Table 5.2c.

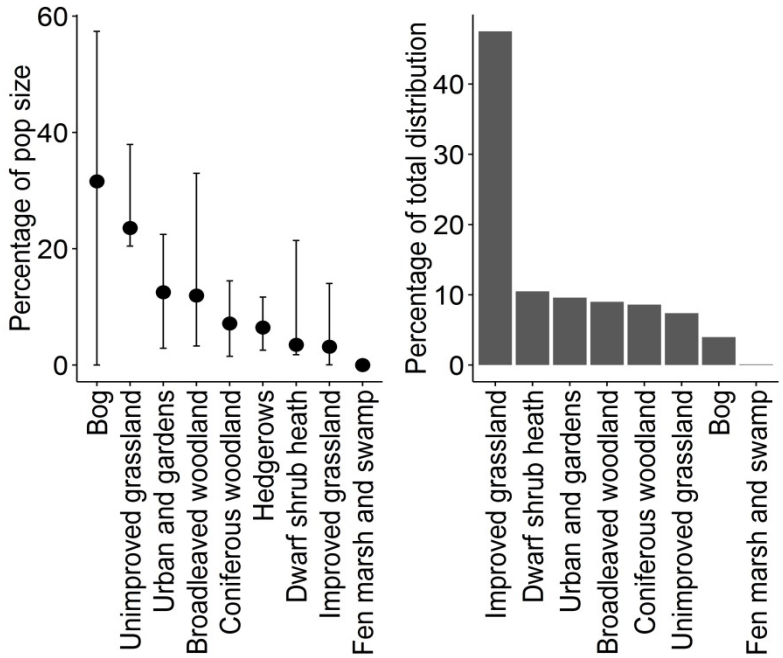


Figure 5.2b Left: The percentage of the total population of common shrews derived from each habitat type. Error bars are obtained by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 5.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Unimproved grassland	Bog	Improved grassland
Location of study sites	0	Estimates from one location		0	
	1	Estimates restricted	1		1
	2	Estimates widespread			
Sample size	0	<10 density estimates		0	0
	1	10-30 density estimates	1		
	2	>30 density estimates			
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**2**	**0**	**1**
		Overall reliability score	**1**		

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated population sizes as 41,700,000 in total, with 26,000,000 in England, 11,500,000 in Scotland and 4,200,000 in Wales. These estimates were made using sparse data on population density, which can be highly variable within each habitat. Similar uncertainties persist in the current estimates, where the confidence limits range from 6 million to 58 million individuals. This level of uncertainty means that no inference can be made about temporal trends.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature search.

Table 5.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995), and trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland*	

* Differences in range are likely to be the result of variable recorder effort.

Drivers of change

Table 5.2e Drivers of population change for the common shrew between 1995 and the present.

Driver	**Mechanism**	**Source**	**Direction of effect**
None quantified, though there are potential impacts of declining invertebrate abundance.	Alterations in farming practice and potentially also use of pesticides.	None	Negative

Data deficiencies

Table 5.2f Areas where further research is required to improve the reliability of population size estimates for the common shrew.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Urban and gardens Improved grassland Fen, marsh and swamp	Density estimates are based on expert opinion.
Density estimates are more than 10 years old.	Hedgerows Bog	Density estimates are from 1997 and 2003, respectively.
No occupancy data.	All habitats except hedgerows	

Future prospects

Table 5.2g An assessment of the future prospects of the common shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline
Range	Stable
Habitat	Decline

5.3 Pygmy shrew *Sorex minutus*

Habitat preferences

The pygmy shrew is found in all terrestrial habitats and shows a preference for areas with dense ground cover, particularly unimproved grasslands (O'Keeffe and Fairley, 1981). It

frequently uses surface burrows through thick vegetation or exploit the abandoned burrows of other species. Abundance is high in wet habitats, such as moorland and blanket bog (Croin-Michielsen, 1966), and also in ditches and on the edges of riparian habitats. Its very high energy requirements, the result of its high surface area:volume ratio, means that it requires habitats with high invertebrate abundance. It may therefore be negatively affected by changes to agricultural practice and/or pesticide use which reduce its prey availability. There is very little research in the recent literature: most records are reported as part of multi-species small mammal studies.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 5.3a. Gaps in the distribution in England and Wales are likely to result from a lack of survey effort, rather than true absences. It is less clear whether larger gaps in Scotland represent true gaps in distribution or are influenced by survey effort. Further surveys are recommended in these areas to increase confidence in the current distribution.

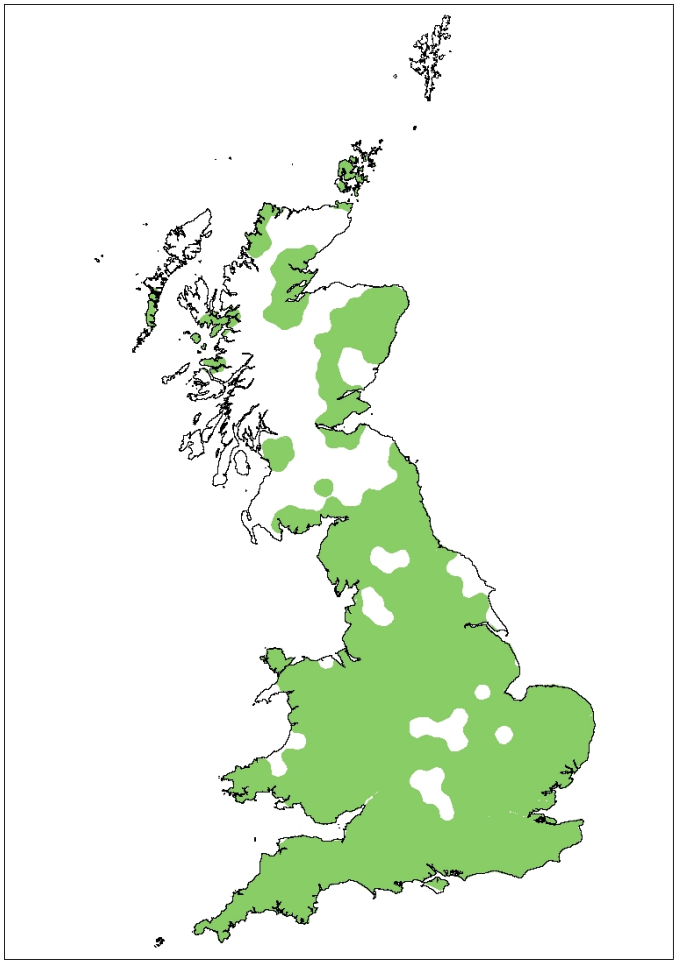


Figure 5.3a Current range of the pygmy shrew in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

As the same animals are likely to use the cropped area and the field margins (hedgerows and buffer strips), we have excluded the estimate for arable fields to avoid counting the same animals twice.

Results

Two papers were identified by the literature search: one reported pre-breeding density estimates, and one contained post-breeding density estimates. Population density estimates are shown in Table 5.3a, and population size estimates are provided in Table 5.3b.

Table 5.3a Median density estimates for pygmy shrews, with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerow length and density within hedgerows are presented as km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Unimproved grassland	823,000	4.80	1.20	4.80	Pernetta (1977)	9	n/a
Improved grassland	5,440,000	0	0	1.0	expert opinion	1	n/a
Bog	333,000	3.0	0	8.50	expert opinion	2	n/a
Broadleaved woodland	1,100,000	1.0	0	3.0	expert opinion	2	n/a
Urban and gardens	1,250,000	0.10	0	5.0	expert opinion	1	n/a
Dwarf shrub heath	870,000	0.10	0	5.0	expert opinion	1	n/a
Fen, marsh and swamp	7,190	0.02	0	2.0	expert opinion	1	n/a
Hedgerows	427,000 (km)	0.1	0.01	30.0	expert opinion	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 5.3b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 5.3a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha) *	Population size	-95%CI	+95%CI
England	6,600,000	[3,690,000]	552,000	27,900,000
Scotland	1,660,000	[1,430,000]	217,000	6,040,000
Wales	1,550,000	[1,170,000]	231,000	4,970,000
Britain	**9,810,000**	**[6,300,000]**	**999,000**	**38,900,000**

* The lengths of hedgerows are 371,000km in England, 9,700km in Scotland and 46,600km in Wales.

Critique

No percentage occupancy data were available; the population size for this species is therefore overestimated. All of the population density estimates except those for unimproved grassland are based on the opinion of two to four experts, depending on habitat. 63% of the population resides in unimproved grasslands, which is supported by nine individual density estimates from one paper (Pernetta, 1977). Stepwise deletion and replacement of each of these nine density estimates reduced population size by 8% in four of the nine instances, but all alternative population sizes fell within the confidence limits of the original. Most of the species' distribution consists of urban areas and gardens (29%), despite population densities being relatively low in this habitat (Table 5.3a). It is likely that at least some of the population found in improved grassland lives primarily within hedgerows. The inclusion of both of these habitat types may have introduced an element of double counting, although only the upper confidence limit would be affected since the median density for improved grassland was estimated to be zero. A reliability assessment is provided in Table 5.3c.

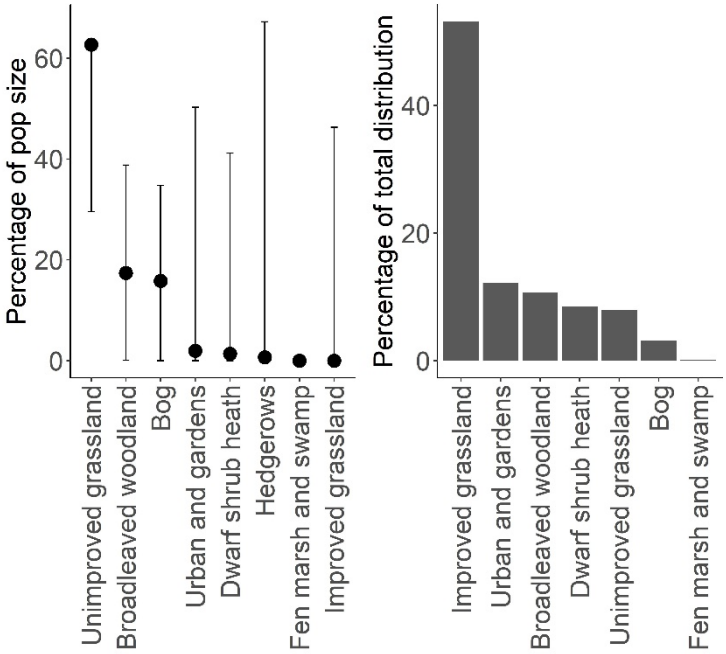


Figure 5.3b Left: The percentage of the total population of pygmy shrews accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 5.3c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**	
			Unimproved grassland	**Urban and gardens**
Location of study sites	0	Estimates from one location	0	
	1	Estimates restricted		1
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	0
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**0**	**1**
		Overall reliability score	**0.5**	

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) reported a total of 8,600,000 pygmy shrews in Britain, with 4,800,000 in England, 2,300,000 in Scotland and 1,500,000 in Wales. These figures are based on the ratio of pygmy shrews to common shrews, which itself was derived from very sparse population density data. Our estimate is based on expert opinion with empirical data for one habitat type only, and so is also highly uncertain. The difference in population size estimates is, therefore, likely to be caused by methodological differences, rather than any true change in population size.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is provided in Table 5.3d.

Table 5.3d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland*	

* Differences in range size are likely to be because of variable recorder effort.

Drivers of change

Table 5.3e Drivers of population change for the pygmy shrew between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
None quantified, though there are potential impacts of declining invertebrate abundance.	Alterations in farming practice and potentially also use of pesticides.	None	Negative

Data deficiencies

Table 5.3f Areas where further research is required to improve the reliability of population size estimates for the pygmy shrew.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Bog Broadleaved woodland Urban and gardens Dwarf shrub heath Fen, marsh and swamp Hedgerows Improved grassland	Density estimates are based on expert opinion.
Limited density estimates for the key habitat.	Unimproved grassland	Median density is based on 9 density estimates.
Density estimates are more than 10 years old.	Unimproved grassland	Density estimates are taken from Pernetta (1977).
No occupancy data.	All habitats	

Future prospects

Table 5.3g An assessment of the future prospects of the pygmy shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline
Range	Stable
Habitat	Decline

5.4 Water shrew *Neomys fodiens*

Habitat preferences

The water shrew is usually associated with aquatic habitats, including rivers, streams, marshes, fens, reed beds, watercress beds and, occasionally, garden ponds (Greenwood et al., 2002). However, it is also found regularly — but infrequently — in non-riparian habitats, including hedgerows, especially those associated with ditches, regardless of whether the features are currently wet (Fiona Mathews, *pers. obs.*). Research on the specific habitat preferences of the species is limited. Greenwood et al. (2002) found that rivers with steep, high banks (45° incline, 1.5m banks) have a higher chance of occupancy, although these characteristics are not essential. A complex relationship was identified between occurrence and vegetation cover: water shrews are found only at sites with trees, and have a higher occurrence where tree cover is sparse. The decrease in occurrence with dense tree cover is thought to be because of the correlated decrease in ground vegetation. High water quality is thought to be an important determinant of occurrence, through both the direct ingestion of pollutants via grooming, and the reduction of prey diversity and abundance. It is currently unclear whether water shrews have been negatively affected by organochlorine pesticides in water. Further testing is required to confirm all habitat associations, which were identified in a single study in the south east of England (Greenwood et al., 2002), and may not represent habitat requirement throughout Britain. Loss of connectivity between habitat patches is likely to have occurred as a consequence of physical features such as dams, weirs, and embankments, as well as human activities such as land draining and the deepening, straightening and widening of channels.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC)
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 5.4a. Gaps in the species' distribution in England and Wales are likely to represent areas lacking survey effort, rather than true absences. It is unclear whether larger gaps in Scotland are true gaps in the distribution or reflect a lack of survey effort.

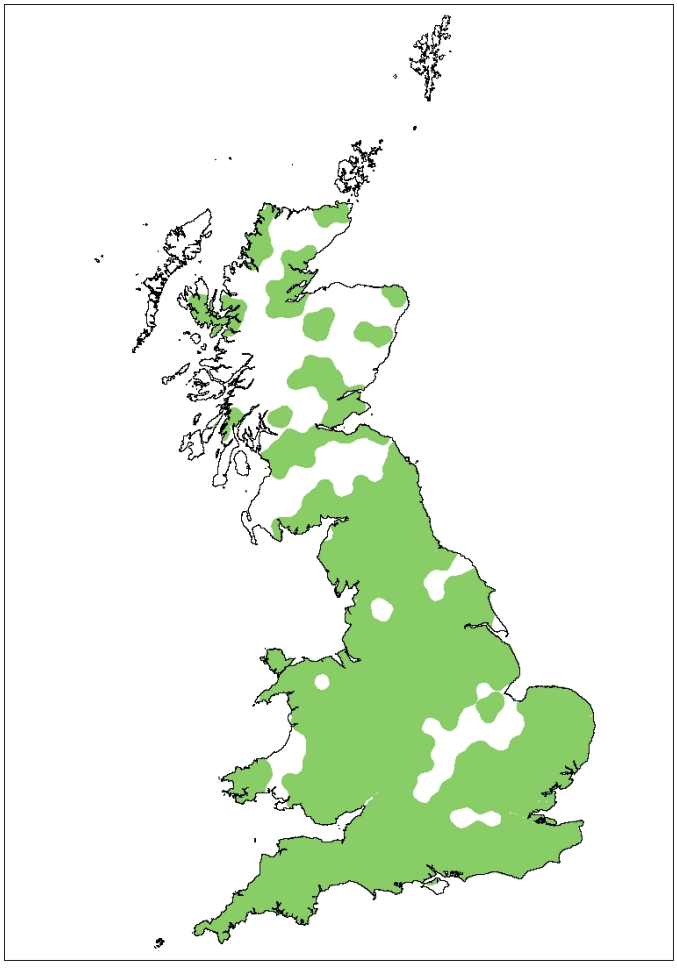


Figure 5.4a Current range of the water shrew in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Harris et al. (1995) estimated population size based on the ratio of common shrews to water shrews, stating that water shrews are known to use a variety of habitats. In the absence of any recent data on population size or density for water shrews, the same method and ratios were used here, although population sizes were adjusted to reflect range size.

First, the density of common shrews in their total range was calculated by dividing the total population size (see Table 5.2b) by area (total range area, not the area of suitable habitat). For the water shrew, density was calculated by dividing the density of common shrews by the ratio of common shrews to water shrews. The density of water shrews was then multiplied by the total range area to give an estimate of total population size for each country.

Results

No papers with pre-breeding population density estimates or trends were identified by the literature search. The density estimate for unimproved grassland was derived from expert opinion ($0.5ha^{-1}$; 95%CI = $0\text{-}1ha^{-1}$). One reference to occupancy for waterways was identified, derived from a national survey carried out in 2004-2005 (Carter and Churchfield, 2006). Surveys took place at 2159 sites across Britain, with signs of water shrews found at 17.4% of sites. As the population size estimate is not habitat-based, occupancy values could not be applied.

Table 5.4a Density of common shrews and water shrews per hectare of their total range. For common shrews, density was calculated by dividing total population size (see Table 5.2b) by area. For water shrews, density was calculated by dividing the density of common shrews by the ratio of common shrews to water shrews.

	Common shrew					Water shrew		
Country	**Area within species' range (ha)**	**Estimate (ha^{-1})**	**-95CI**	**+95CI**	**Ratio***	**Estimate (ha^{-1})**	**-95CI**	**+95CI**
England	1,280,000	8.59	2.75	23.1	22.1 : 1	0.39	0.12	1.04
Scotland	529,000	14.5	3.74	43.3	31.7 : 1	0.46	0.12	1.36
Wales	194,000	12.0	5.20	31.5	15.3 : 1	0.78	0.34	2.06

*Ratio of common shrew to water shrew as derived from Harris et al (1995).

Table 5.4b Total population size estimates, with 95% confidence intervals, for England, Scotland, Wales, and the whole of Britain. Values were obtained by multiplying the range area with the density estimates for water shrews in Table 5.4a. The length of waterways within the species' range is provided, but was not used in the current analysis.

Country	Area within species' range (ha)	Population size	-95%CI	+95%CI
England	11,780,000	[458,000]	147,000	1,228,000
Scotland	2,580,000	[118,000]	30,000	353,000
Wales	1,750,000	[137,000]	60,000	361,000
Britain	**16,115,000**	**[714,000]**	**237,000**	**1,942,000**

Critique

The population size estimate for the water shrew is based on the ratio with the common shrew. It is therefore subject to the same errors as the common shrew population size estimate, in addition to uncertainties in the ratio. The use of a ratio as a method is questionable because of the differing habitat preferences of the species.

Fifty-six percent of the population size for common shrews (on which the water shrew estimate is based) is derived from unimproved grassland and bog. The average density estimate for common shrews in unimproved grassland is supported by 27 replicate density estimates, and the estimate for bog is based on data from Shore and Mackenzie (1993), which was provided by the authors. Reliability scores of two and zero were applied to the population size estimates for these habitats, respectively, and the overall confidence limits for common shrews are wide (6 million to 58 million), reflecting the uncertainties associated with all stages of the analysis. The ratio of the common shrew to the water shrew was based on bird of prey pellet analysis, and bottle and trap samples from England, Scotland and Wales (see Tables 5-8 in Harris et al., 1995). Although the sample size for these ratios is high (42 papers and one value derived from expert opinion), they do not take into account the high variation in population density and patchy distribution of water shrews though to occur in the British population (Harris et al., 1995).

Harris et al. (1995) estimated the population size of water shrews to be 1,900,000, with 1,200,000 in England, 400,000 in Scotland and 300,000 in Wales. These estimates, however, are not adjusted to take into account the smaller distribution of the water shrew

compared to the common shrew. Instead, in the current review, the population size in Britain was computed by dividing the population size of the common shrew by the ratio of the common shrews to the water shrew per country. Reassessment of the data from Harris et al. (1995) using the method presented here suggests a total population size in Britain of 1,500,000 ((common shrew population size/current common shrew distribution area/ratio of common shrew to water shrew) * area of water shrew distribution), although this method assumes that the species' range has remained constant since 1995.

Table 5.4c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All habitats
Location of study sites	0	Estimates from one location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 density estimates	0
	1	10-30 density estimates	
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**0**
		Overall reliability score	**0**

Changes through time

Comparison to Harris et al. (1995)

Population sizes were estimated by Harris et al. (1995) to be 1,900,000 in Great Britain, comprising 1,200,000 in England, 400,000 in Scotland and 300,000 in Wales. The current review applied the same approach as Harris et al. (1995), using the ratio of the common shrew to the water shrew. Both the current and previous estimates for the common shrew are uncertain, being based on few density estimates, and share the same sources of error. The population size of 1,500,000 estimated by Harris et al. (1995), however, falls within the confidence limits of the current estimate. Further surveys are needed to increase confidence in the estimated population sizes and allow for an assessment of trends.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature search. A summary of trends in population size and range is provided in Table 5.4d.

Table 5.4d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	England Wales*	Scotland		

* Increase in range may be the result of increased survey effort.

Drivers of change

Table 5.4e Drivers of population change for the water shrew between 1995 and the present.

Driver	**Mechanism**	**Source**	**Direction of effect**
Habitat availability.	Bank clearance and modification may destroy burrows and alter water supplies. Effects of wide-scale alterations unknown.	Howie and Stokes (2003)	Negative
None quantified, though there are potential impacts of declining invertebrate abundance.	Alterations in farming practice and potentially also use of pesticides.	None	Negative

Data deficiencies

Table 5.4f Areas where further research is required to improve the reliability of population size estimates for the water shrew.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	All habitats	Population size estimates are based on ratios with wood mice.
No occupancy data.	All habitats	

Future prospects

Table 5.4g An assessment of the future prospects of the water shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline

5.5 Lesser white-toothed shrew *Crocidura suaveolens*

Habitat preferences

The lesser white-toothed shrew is the only shrew species present on the Isles of Scilly, where it is found on all of the larger islands. It makes use of all habitat types, as long as adequate cover can be found. Common in habitats with tall vegetation such as bracken, hedgerows and woodlands (Harris and Yalden, 2008), it is also reported to use foreshores of the Isles of Scilly (Temple and Morris, 1997). Males have larger home ranges than females, although both are less than 100m in length (Harris and Yalden, 2008).

6 LAGOMORPHA

6.1 European rabbit *Oryctolagus cuniculus*

Habitat preferences

The rabbit is found in a wide variety of habitats, but prefers those with short grass such as improved grasslands or arable areas. Peaks in population size are found in areas with sandy soils and chalk, as opposed to clay soils (Cowan, 1991; Harris et al., 1995).

Rabbit density is positively associated with livestock grazing, owing to the higher nitrogen content of grazed swards (Iason et al., 2002; Bakker et al., 2005; Lush et al., 2014). More specifically, the rabbit prefers shorter grass swards (Smith et al., 2005; Petrovan et al., 2011a) with low plant diversity (i.e. intensively grazed pasture; Lush et al., 2014), and with predator control measures in place. Livestock production in the UK is in decline (UK National Ecosystem UK National Ecosystem Assessment, 2011). However, several features of less intensively managed landscapes, such as the presence of field margins, hedgerows, and woodland, are also beneficial (Trout et al., 2000; Petrovan et al., 2011a). These features presumably offer the necessary cover to escape predators (Iason et al., 2002), while still providing high quality forage (Bakker et al., 2005).

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: NT.).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

The status of rabbits as ‘near threatened’ on the IUCN Red List is based on the species’ native range, which does not include Britain.

Species' distribution

A distribution map is presented in Figure 6.1a. Gaps in the species' distribution in Scotland are likely to represent a lack of survey effort, rather than true absences.

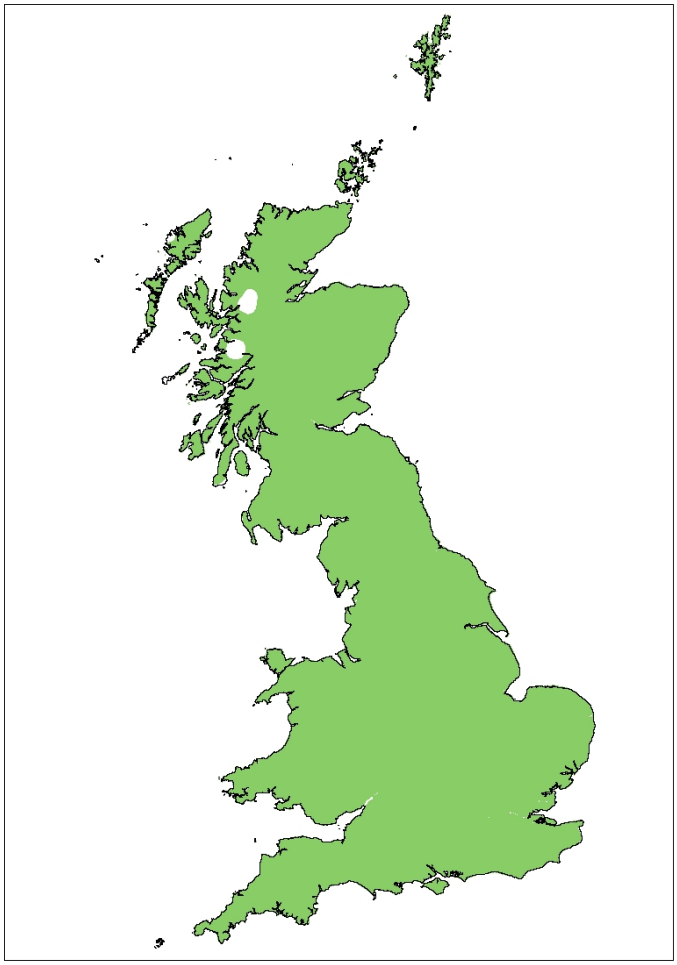


Figure 6.1a Current range of the European rabbit in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

Five papers were identified by the literature search: one contained estimates of pre-breeding population density; two provided post-breeding estimates; one provided an index of rabbit abundance; and one provided evidence of a temporal trend.

Population density estimates per habitat are provided in Table 6.1a, and total population size estimates in Table 6.1b.

Table 6.1a Median density estimates with 95% confidence intervals for European rabbits, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^2)	-95%CI	+95%CI	Source*	n**	%Occ†
Improved grassland	73,100	48.3	26.3	600	Petrovan et al. (2011a) ††	7	n/a
Arable and horticulture	62,600	250	-	-	Harris et al. (1995)	1	n/a
Broadleaved woodland	13,100	200	-	-	Harris et al. (1995)	1	n/a
Coniferous woodland	14,400	200	-	-	Harris et al. (1995)	1	n/a
Dwarf shrub heath	19,700	250	-	-	Harris et al. (1995)	1	n/a
Unimproved grassland	12,400	500	-	-	Harris et al. (1995)	1	n/a
Supra-littoral rock	32	500	-	-	Harris et al. (1995)	1	n/a
Supra-littoral sediment	300	500	-	-	Harris et al. (1995)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.
†† Raw data were supplied by the authors.

Table 6.1b Area of suitable habitat (not adjusted for occupancy) within the species' range. Values were obtained by multiplying population density estimates with the area of habitat within the species' distribution. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	115,000	[21,300,000]	-	-
Scotland	61,300	[11,800,000]	-	-
Wales	19,100	[2,910,000]	-	-
Britain	**196,000**	**[36,000,000]**	**-**	**-**

Critique

No percentage occupancy data were available, so the population size for this species is overestimated. 42% of the population is attributed to arable and horticultural land, for which the density estimate is taken from Harris et al. (1995). Most of the land within the species' range consists of arable and horticulture (32%) and improved grassland (37%).

The density estimates derived from Harris et al. (1995), and hence the overall population size estimates, are somewhat at odds with the known preference of rabbits for areas with short grass swards and low plant diversity, such as improved grassland (Lush et al., 2014). For example, the densities given for unimproved grassland and coniferous woodland are 500 rabbits km^{-2} and 200 rabbits km^{-2} respectively, whereas the recent estimate for improved grassland is only 48 rabbits km^{-2}. However, given the extreme variability in rabbit abundance, a large sampling effort is required to produce robust evidence on median densities with reasonable precision. Therefore, using a single central estimate for each study is likely to introduce considerable error. The density estimate for improved grassland is based on a relatively small amount of data (one paper with seven replicates; Petrovan et al., 2011a), resulting in 95% confidence limits of $30km^{-2}$ to $60km^{-2}$. These densities were estimated using spotlight counts, which may only represent ~60% of the total number of rabbits within the unit area (Poole et al., 2003; Petrovan et al., 2011a), although this percentage may vary greatly depending on the area under study. Further data would provide a more robust estimate, but it is likely that a density at the upper confidence limit is more representative for this habitat type. The estimate for improved grassland was $250km^{-2}$ in Harris et al. (1995), which though considerably higher, falls within the confidence limits.

Estimates from Harris et al. (1995) were based on the authors' adjustment of over-wintering population estimates from high density areas.

Factors such as outbreaks of myxomatosis and rabbit haemorrhagic disease have severe local impacts (Petrovan et al., 2011b), but these are rather poorly understood on national scales. The long-term impacts of these diseases on populations are also unclear (beyond anecdotal evidence of regional recoveries from previous population crashes), making it particularly difficult to extrapolate from historical evidence on habitat-specific densities. In addition, rabbit populations are inherently highly variable, even in the absence of disease, so there is considerable uncertainty in density estimates both within and between habitat types. The application of a single median density (current method) or single adjusted density (Harris et al., 1995), particularly where data are limited, may not, therefore, result in a reliable population size estimate. A stratified survey approach, using different geographical regions and habitat types, would improve the current estimates.

Stepwise deletion and replacement of each of the seven replicates for improved grassland resulted in two alternative population size estimates, both of which differed from the original by 29% (46,000,000), although the lack of confidence limits for the original population size estimates means that the significance of this cannot be formally assessed.

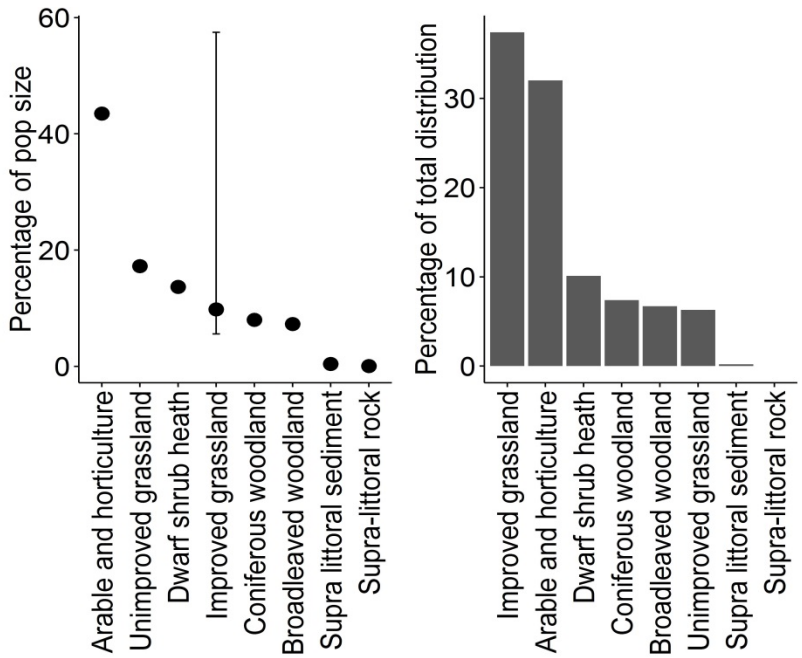


Figure 6.1b Left: The percentage of the total population of European rabbits accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 6.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Arable and horticulture	Improved grassland
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	0
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**1**
		Overall reliability score	**1**	

Changes through time

Comparison to Harris et al. (1995)

Population size estimates in Harris et al. (1995) were 37,500,000; 24,500,000 in England, 9,500,000 in Scotland and 3,500,000 in Wales. The density estimates for the majority of habitats were taken from Harris et al. (1995), so a comparison of population sizes is limited to differences in range size and habitat availability.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a 9% reduction in population size. The lack of confidence limits around the current estimate means that the significance of this reduction is unclear.

Other evidence of changes through time

The National Gamebag Census reported a decrease of 24% (95%CI 45% decrease to 4% increase) in the number of rabbits culled between 1995 and 2014 in the UK. The NGC survey, however, does not account for effort, so it may not represent a true decline in population size.

The Breeding Bird Survey inferred a population decrease of 48% (95% CI 56%-33%) between 1995 and 2012 in the UK (Wright et al., 2014). Two experts also suggested that the population may have declined in size, but highlighted that the situation is complex because of the wide range of rabbit densities present in different areas, and uncertainties about the rate of change.

Table 6.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		All countries*		
	Data deficient				

* Based on BBS trend data and expert opinion.

Drivers of change

Table 6.1e Drivers of population change for the European rabbit between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Disease/pesticides.	Recovery from myxomatosis since the 1950s epidemic.	Petrovan et al. (2011a)	Positive
	The impact of rabbit haemorrhagic disease.	Petrovan et al. (2011a)	Negative
Management (control).	Rabbits are culled where damage is caused to agriculture. This driver is, however, complex, and culling effort may have been reduced owing to lower demand for rabbit meat or fur.		Negative

Data deficiencies

Table 6.1f Areas where further research is required to improve the reliability of population size estimates for the European rabbit.

Data deficiencies	Habitat	Details
Density estimates are >10 years old.	All habitats except improved grassland	Density estimates were taken from Harris et al. (1995).
Limited density estimates.	Improved grassland	One paper contained density estimates with fewer than 10 replicates. Sensitivity analysis suggests that the population estimate is highly dependent on two data points.
Density estimates do not represent within-habitat variability.	All habitats except improved grassland	No range or confidence limits were available for the density estimates.
Density estimates do not represent within-habitat variability.	All other habitats	It was not possible to calculate confidence intervals owing to lack of data. Rabbit populations are known to be highly variable within and between habitats.
No occupancy data.	All habitats	

Future prospects

Table 6.1g An assessment of the future prospects of the European rabbit, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline*
Range	Stable
Habitat	Decline

* Note that much of the expected decline is because of disease impacts, and there is uncertainty about future epidemiological patterns and population impacts.

6.2 **Brown hare** *Lepus europaeus*

Habitat preferences

The brown hare is found mainly in lowland arable or pastoral land. It is frequently found in open areas, where it uses its swift running speed to evade predators, but it also requires shelter for resting and breeding. Its abundance is positively associated with habitat and plant species diversity, the presence of hedgerows, and unfarmed habitat (Tapper and Barnes, 1986; Smith et al., 2004; Lush et al., 2014), and negatively associated with grazing intensity. The brown hare selects habitat based on structure, rather than the availability of nutrients, which is thought to result from the need for cover from predators and for surface resting sites (Smith et al., 2004; Lush et al., 2014). Population density may be adversely affected by shooting, competition with farm livestock for food, and the intensification of agricultural practices (Hewson and Hinge, 1990).

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

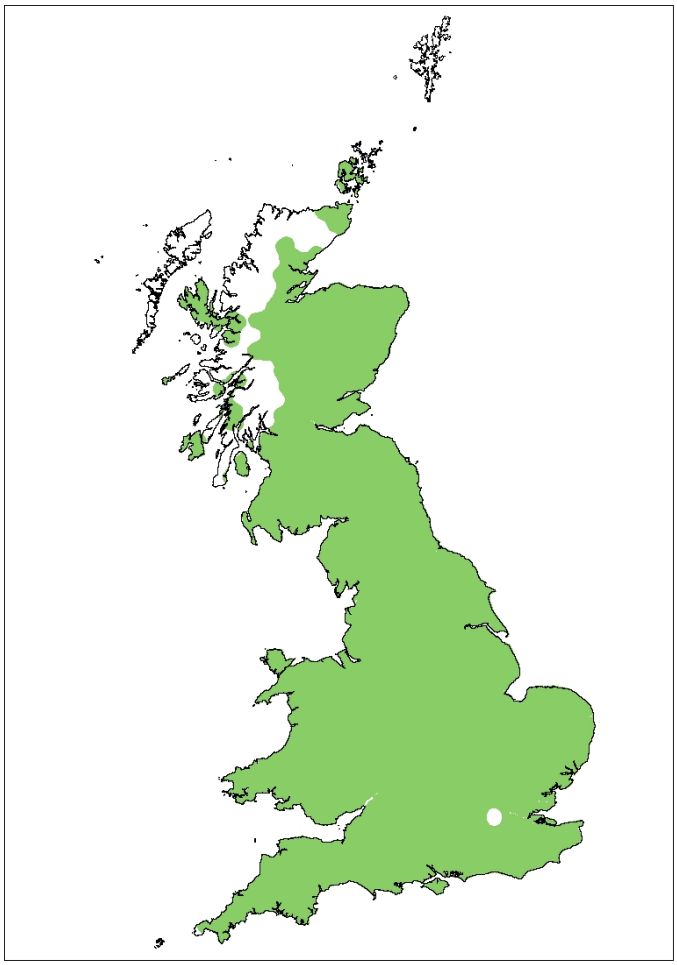


Figure 6.2a Current range of the brown hare in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Population size estimates were based on density data for arable habitat and improved grassland only. Although brown hares may be found elsewhere, arable land and improved grassland constitute their core habitat. The use of occurrence data from other habitats was judged likely to introduce double counting, because most animals observed here would already be accounted for in the arable and grassland data.

Results

Twelve papers and one NGO report were found for the brown hare, where seven contained pre-breeding estimates of population size, five contained post-breeding estimates or temporal trend data, and one contained occupancy data (Hutchings and Harris, 1996). Percentage occupancy was measured as the number of positive 1km survey squares (456 of 738); habitat-specific occupancy values were not available so this percentage was applied to all habitats. Population density estimates are provided in Table 6.2a, and population size estimates in Table 6.2b.

Table 6.2a Median density estimates with 95% confidence intervals for brown hares, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Arable and horticulture	62,500	11.0	8.25	29.6	Bradshaw (1993)	2	61.8
					Heydon et al. (2000)	6	
					Hutchings and Harris (1996)	2	
						2	
					Rothschild and Marsh (1956)	2	
					Temple et al. (2000)		
Improved grassland	68,800	3.63	2.55	26	Bradshaw (1993)	2	61.8
					Heydon et al. (2000)	1	
					Hutchings and Harris (1996)	2	
						5	
					Parrott et al. (2012)	2	
					Petrovan et al. (2011b)	2	
					Rothschild and Marsh (1956)	2	
					Temple et al. (2000)		

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 6.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 6.2a with the area of habitat within the species' distribution, and adjusting for occupancy.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	93,900	454,000	336,000	1,480,000
Scotland	24,300	87,700	64,000	342,000
Wales	13,100	37,300	26,800	171,000
Britain	**131,000**	**579,000**	**427,000**	**1,990,000**

Critique

The population estimate for brown hares is based on densities in arable and horticultural habitats (73%), and in improved grassland (27%; Figure 6.2b). Twelve and 14 individual estimates, respectively, were obtained for these habitat types. Although population size was adjusted to reflect occupancy, these data are not habitat-specific; the brown hare population is also patchily distributed throughout the country, with 19% of occupied squares found in three counties (Hutchings and Harris, 1996). Percentage occupancy data are therefore likely to introduce inaccuracies when applied to the whole country. A reliability assessment is provided in Table 6.2c.

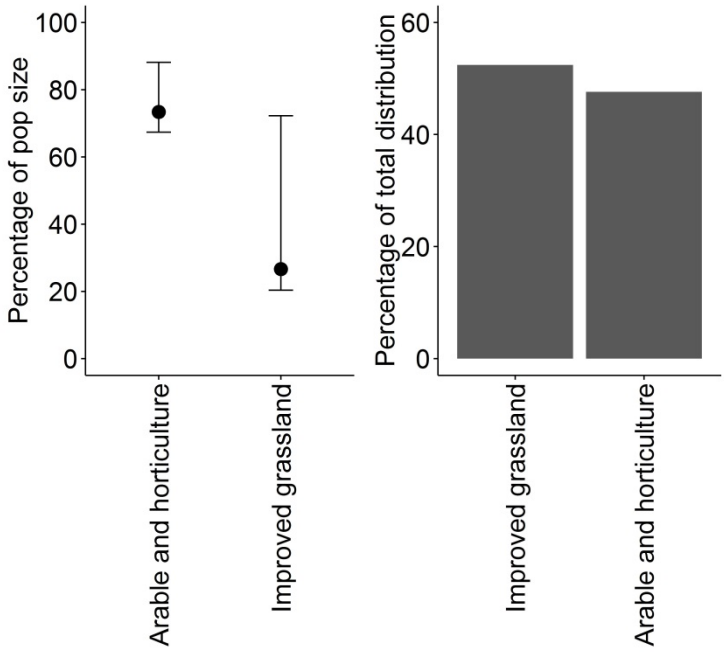


Figure 6.2b Left: The percentage of the total population of brown hares accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 6.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Arable and horticulture	Improved grassland
Location of study sites	0	Estimates from one location		
	1	Estimates restricted*	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates		
	1	10-30 density estimates	1	1
	2	>30 density estimates		
Occupancy data available?	0	No		
	1	Yes	1	1
		Habitat score	**3**	**3**
		Overall reliability score	**3**	

***** Although studies covered a wide geographical range, they did not capture the extremely large variability in density and occupancy known to occur between eastern and western counties of England. Therefore, a score of 1 has been allocated to this factor.

Changes through time

Comparison to Harris et al. (1995)

In Harris et al. (1995), population size estimates were 817,000 in total, comprising of 572,000 in England, 187,250 in Scotland, and 58,000 in Wales. These estimates were provided by the first National Brown Hare Survey, which was later published by Hutchings and Harris (1996). These figures were derived from extensive surveys and subsequent population density estimates, grouped by land class rather than habitat type, and included occupancy data by incorporating sampled areas with no detection of hares. By contrast, the current estimate is based on the two main habitats for brown hares only (see 'Species-specific methods').

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size only 5% different from the original (and within the original confidence limits). These methodological differences are therefore unlikely to affect comparisons between the two reviews materially.

Although the population size estimated by Harris et al. (1995) is higher than the current estimate, it falls within the confidence limits. This fact, coupled with the differences in methodology and habitat associations, means it is difficult to make direct comparisons between the current review and that of Harris et al. (1995).

Other evidence of changes through time

The National Gamebag Census reports a 38% (95%CI = 3%-76%) increase in brown hares shot for game from 1995 to 2009. This trend may not, however, represent a change in the absolute population size, as survey effort may have differed between years.

Following the first National Brown Hare Survey, which estimated brown hare population size to be 817,000 in 1991-1993 (Harris et al., 1995; Hutchings and Harris, 1996), a second survey in 1997-1999 (Temple et al., 2000) estimated the population to be 752,608 (95%CI = 714,911-790,305). As this decline is relatively small, its significance is difficult to determine,

particularly given the lack of confidence limits for the former estimate. Even if relatively tight confidence limits are assumed, the difference is unlikely to be significant. A summary of trends in population size and range is provided in Table 6.2d.

Table 6.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries		

Drivers of change

Table 6.2e Drivers of population change for the brown hare between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Changes to agricultural practices, particularly the switch towards winter wheat, may reduce food and shelter opportunities. Competition with livestock for food.	Hewson and Hinge (1990)	Negative
Management (control).	Suppression of the population during specified times of year.	Tapper and Parsons (1984)	Negative
Climate change.	Alteration of agricultural practice, particularly the switch away from arable crops in eastern counties, may be detrimental to hare populations, although potentially offset by increases in arable production in the north and west.	Fezzi et al. (2014)	Negative

Data deficiencies

Table 6.2f Areas where further research is required to improve the reliability of population size estimates for the brown hare.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	Arable and horticulture	The most recent density estimate is from Heydon et al. (2000).
Managed populations.	Arable and horticulture	Population management is not taken into account in the population size estimate.
Occupancy and density estimates.	All	It is not currently possible to adjust for known variability in density and occupancy between eastern and western counties owing to a lack of data.

Future prospects

Table 6.2g An assessment of the future prospects of the brown hare, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Decline

Future prospects

Table 7.5g An assessment of the future prospects of the edible dormouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase**
Range	Increase*
Habitat	Increase

* Very slow increase owing to slow reproduction, lack of dispersal behaviour, and habitat barriers.
** Increase in population size is inferred from an increase in range.

7.6 Bank vole *Myodes glareolus*

Habitat preferences

The bank vole is found in a variety of habitats, including hedgerows, conifer plantations and road verges, but shows a strong preference for mature broadleaved and mixed woodland (Flowerdew et al., 2004). The diet comprises fruits, seeds and leaves from broadleaved trees, although other food sources such as flowers, grasses and moss are taken opportunistically. Unlike populations in mainland Europe, the bank vole forms a high proportion of its winter diet in Britain from dead leaves (Hansson, 1985).

Limited recent research is available on the factors affecting the population density of bank voles, although abundance is positively associated with the quality and size of woodlands. The species requires dense ground vegetation (Fernando et al., 1994). It is found frequently in field margins and hedgerows, which can support large resident populations (Gelling et al., 2007), but only rarely in arable fields (Harris and Yalden, 2008). Arable habitats were therefore excluded from the population estimates. There is no evidence of multi-annual cycles for bank voles in Britain (Flowerdew et al., 2004).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.6a. Gaps in the distribution in England and Wales are likely to represent areas lacking survey effort, rather than true absences. It is unclear whether the larger gaps in Scotland reflect a lack of recorder effort or true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

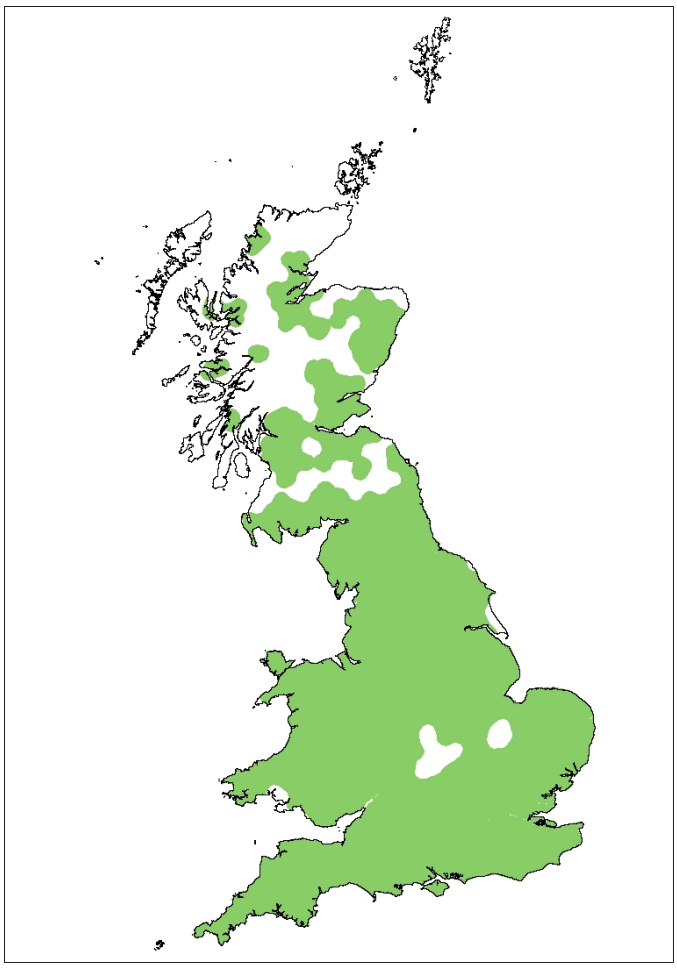


Figure 7.6a Current range of the bank vole in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

The literature search (using both *Myodes glareolus* and *Clethrionomys glareolus* as species names) identified 63 papers. Fourteen provided information on population size or distribution; of these, six gave pre-breeding estimates of population density, and one contained percentage occupancy for hedgerows (Gelling et al., 2007). The remaining papers reported post-breeding estimates, assessed the relative effects of environmental variables on population size, or gave distribution data only. Population density estimates by habitat are shown in Table 7.6a, and total population size estimates in Table 7.6b.

Table 7.6a Median density estimates for bank voles with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerows are divided according to whether they were managed through an agri-environment scheme (AES and non-AES). Hedgerow length and density within hedgerows are shown in km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	1,200,000	9	6	10	Flowerdew et al. (2004), Hare (2009)	147 2	n/a
Urban areas and gardens	1,320,000	4.5	3.0	7	expert opinion	2	n/a
Coniferous woodland	944,000	5	3.5	12	expert opinion	2	n/a
Dwarf shrub heath	966,000	1	0.05	3	expert opinion	1	n/a
Fen, marsh and swamp	6,900	30	0	50	expert opinion	1	n/a
Improved grassland	6,070,000	0.1	0	1	expert opinion	2	n/a
Unimproved grassland	900,000	1	0	3	expert opinion	3	n/a
Hedgerows (AES)	139,000 (km)	10.5	0	26	Shore et al. (2005) Broughton et al. (2014) Kotzageorgis and Mason (1997)	24 12 9	96%
Hedgerows (Non-AES)	311,000 (km)	6.3	1.2	8			

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.6b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.6a with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	7,120,000	19,100,000	10,400,000	35,600,000
Scotland	2,520,000	5,390,000	3,130,000	11,900,000
Wales	1,770,000	2,930,000	1,560,000	6,560,000
Britain	**11,400,000**	**27,400,000**	**15,100,000**	**54,100,000**

* The lengths of hedgerows are 386,000km in England, 14,000km in Scotland, and 50,400km in Wales.

Critique

Percentage occupancy data were not available for most habitats; the population size is therefore overestimated. Most of the population estimate is derived from broadleaved woodland (39%). Yet broadleaved woodland forms a low proportion of the land cover within the species' range (Table 7.6a), and its importance is therefore largely a consequence of high density estimates relative to other habitats. Many of the density estimates for these other habitats were derived from expert opinion, highlighting the need for detailed surveys in habitats lacking empirical data.

Just over half of the habitat within range is improved grassland (Table 7.6a). It is estimated that this large habitat area supports only 1% of total bank vole population because of low population density ($0.1ha^{-1}$). This estimate is again the average of values provided by two expert opinions, and validation of the estimated values would improve future calculations of population size. Despite the low suitability of improved grassland and arable land for bank voles, a significant proportion of the population is found in hedgerows in these areas. It is possible that many individuals captured within improved grassland reside in hedgerows, and therefore the inclusion of both these habitats could have slightly overestimated the population. (Arable land was excluded.) A reliability assessment is provided in Table 7.6c. For density estimates based on expert opinion, a conservative score of 1 has been applied to the 'location of study sites' section.

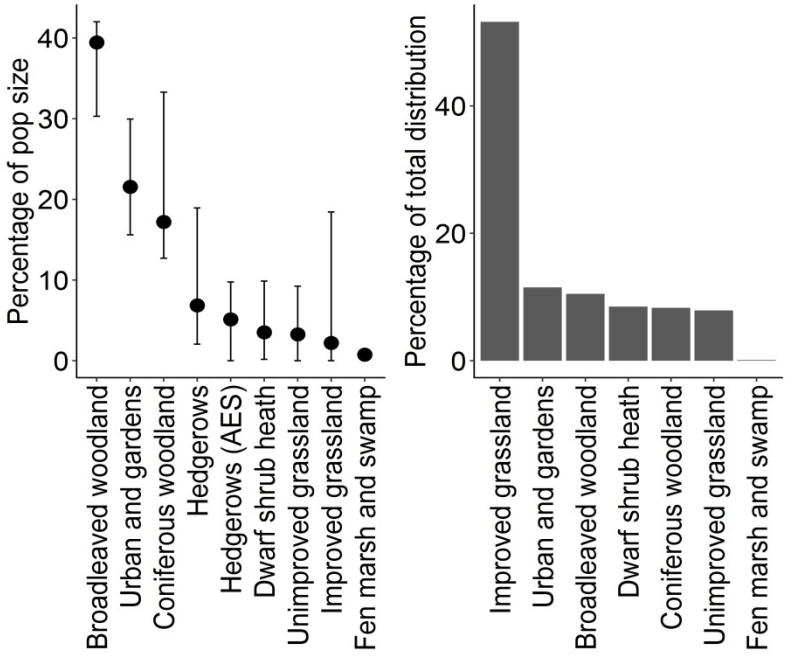


Figure 7.6b Left: The percentage of the total bank vole population accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.6c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Broadleaved woodland	**Unimproved grassland**	**Improved grassland**
Location of study sites	0	Estimates from one location			
	1	Estimates restricted		1	1
	2	Estimates widespread	1		
Sample size	0	<10 population density estimates		0	0
	1	10-30 population density estimates			
	2	>30 population density estimates	2		
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**3**	**1**	**1**
		Overall reliability score	**1.7**		

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) reported a total population size of 23,000,000, comprised of 17,750,000 in England, 3,500,000 in Scotland and 1,750,000 in Wales. Those values fall within the confidence limits of our estimates here, except in Scotland for which current estimates are somewhat higher. Population sizes were calculated by Harris et al. (1995) from density estimates for hedgerows, woodland, scrub and bracken only, where the density per habitat type was assigned using a combination of empirical data and expert opinion. The methods to estimate current population size are, therefore, similar, although the difference in selected habitats and use of expert opinion mean that comparisons should be made with caution.

Nationally, there are changes between the two reviews in the estimated availability of key habitats, generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size only 5% different from the original (and within the original confidence limits). These differences in assumed habitat areas therefore do not materially affect the comparison of population sizes between the reviews.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is shown inTable 7.6d.

Table 7.6d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland*	

Drivers of change

Table 7.6e Drivers of population change for bank voles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Unknown.			

Data deficiencies

Table 7.6f Areas where further research is required to improve the reliability of population size estimates of bank voles.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Fen, marsh, swamp, heathland, grassland, different types of conifer forest, urban, suburban.	Density estimates are currently based on expert opinion.
No occupancy data.	All habitats except hedgerows.	

Future prospects

Table 7.6g An assessment of the future prospects of the bank vole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

7.7 Field vole *Microtus agrestis*

Habitat preferences

The field vole is most abundant in rough low-productivity grassland under low intensity management and untreated by artificial fertilisers. Long tussocky grass allows for the formation of runs and nests (Gelling et al., 2007), and provides protection from aerial predators. The species' density is negatively associated with grazing pressure, but low intensity grazing, particularly by sheep, may be beneficial as it leads to more diverse vegetation structure (Schmidt et al., 2005). The field vole may also live in marginal habitats: it can occupy open grassy patches within fragmented woodlands, as well as moorlands, at low densities (Bellamy et al., 2000; Tattersall et al., 2000), and marginal rough grasslands may support high densities.

Linear habitats, such as hedgerows, are becoming an increasingly important because of habitat fragmentation and the loss of tussocky grasslands (Tattersall et al., 2002). Not only can hedgerows provide useful corridors, but the vegetation in hedgerow bottoms can also provide the sole habitat for the species, especially within pastoral landscapes with typical grazing intensities (Gelling et al., 2007).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 7.7a. Gaps in the species' distribution throughout the mainland are likely to represent areas lacking survey effort, rather than true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

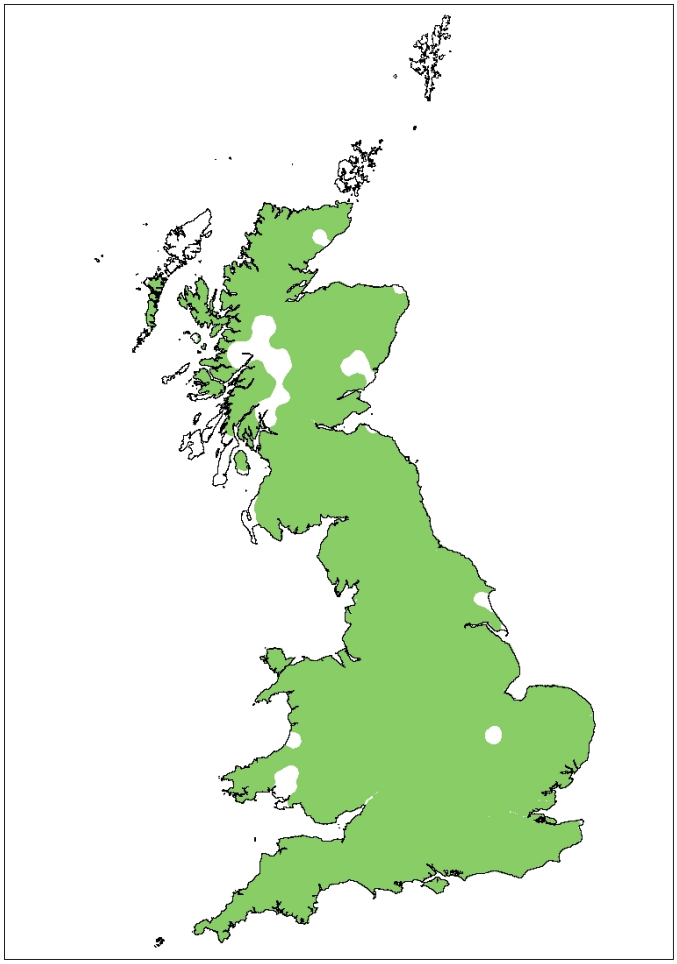


Figure 7.7a Current range of the field vole in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Field voles use long tussocky grass as their primary habitat, so it was assumed that animals within both arable and improved grassland would be transient individuals that would be largely accounted for by the estimates for hedgerows. Arable and improved grassland habitats were therefore not included in the calculation of population size. Annual and multi-annual cycles (three to four year periodicity) in population size are known in Britain, but non-cyclic populations are also present in different areas (Lambin, 2008). These differences make it difficult to assess the status of the population as a whole, or even the stage of the cycle reached at any given time. Therefore, whenever temporal data were available, troughs in population size were used as the best estimate of pre-breeding density, even if these were not obtained in the spring (Lambin et al., 2000; Loughran, 2006).

Results

Nine papers were identified by the literature search. Of these, four contained pre-breeding estimates of population size, four reported post-breeding estimates of population size, and one examined the effect of habitat variables on relative population size. Habitat-specific population densities and total population size estimates are presented in Table 7.7a and Table 7.7b, respectively.

Table 7.7a Median density estimates for field voles with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerows are divided according to whether they were managed through an agri-environment scheme (AES and non-AES). Hedgerow lengths are in km, and animal density within hedgerows is in km^{-1}.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Unimproved grassland	1,100,000	48	33.33	52.5	(Lambin et al., 2000) Flowerdew et al. (2004) Loughran (2006)	11 2 5	n/a
Bog	757,000	1	0	1	expert opinion	2	n/a
Broadleaved woodland	1,270,000	1	0	2	expert opinion	2	n/a
Urban and gardens	1,360,000	1.5	0.05	2.5	expert opinion	2	n/a
Coniferous woodland	1,310,000	1	0	1.5	expert opinion	2	n/a
Fen, marsh and swamp	8,570	5	0	12	expert opinion	1	n/a
Hedgerows (AES)	143,000 (km)	7.5	2	52	Broughton et al. (2014)	12	n/a
Hedgerows (Non-AES)	319,000 (km)	2	0	20			

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.7b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.7a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	3,090,000	28,600,000	16,900,000	44,000,000
Scotland	2,150,000	21,500,000	13,600,000	24,500,000
Wales	561,000	9,760,000	6,430,000	11,800,000
Britain	**5,810,000**	**59,900,000**	**37,000,000**	**80,300,000**

* The lengths of hedgerows are 396,000km in England, 18,700km in Scotland, and 47,300km in Wales.

Critique

No percentage occupancy data were available; the population size is therefore overestimated. Most of the estimated population (88%) is derived from unimproved grassland (Figure 7.7b). The population density used for this habitat is based on 18 sites reported in three papers. Unimproved grassland accounts for 19% of the habitat found within the species' distribution. Coniferous woodland, broadleaved woodland and urban areas form most of the remaining area (67%), although the densities within these habitats are thought to be low (Table 7.7a). Improved grassland was excluded from the analysis on the grounds that it offers poor habitat for field voles, and individuals using this environment would be accounted for by the hedgerow density estimates.

Grassland habitats have spectrally similar profiles in remotely sensed datasets, so areas of rough grassland may have been misclassified as improved grassland, and vice versa, in the LCM2007 dataset. The area given for rough grassland may therefore be inaccurate. We also grouped improved, neutral, acid and calcareous grasslands together for the analysis, combining those habitats most likely to be mistaken for each other but which are also functionally similar. Some of this 'improved grassland' grouping may support a low density of field voles, particularly if grazing intensity is low (Schmidt et al., 2005), but the habitat has been excluded from this analysis. A reliability assessment is provided in Table 7.7c.

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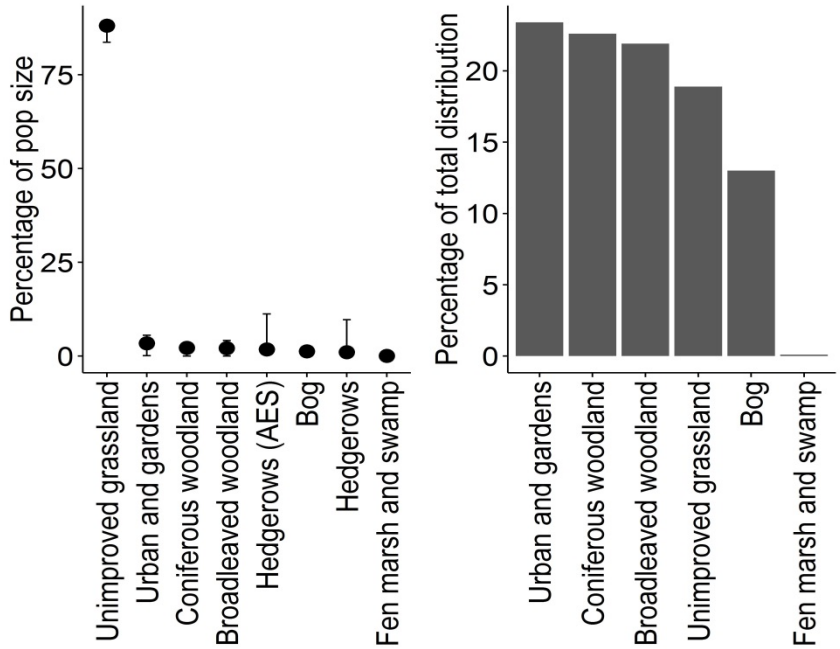


Figure 7.7b Left: The percentage of the total population of field voles accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.7c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Unimproved grassland
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	
	1	10-30 population density estimates	1
	2	>30 population density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**2**
		Overall reliability score	**2**

Changes through time

Comparison to Harris et al. (1995)

The population size estimate in Harris et al. (1995) was 75,000,000. This estimate was based on the ratio of field voles to other small mammals in a range of samples, including live traps and owl pellets. Values of 1.9 field voles per wood mouse and 1.8 field voles per common shrew were derived across all samples. As this method is not based on the area of suitable habitat within the species' distribution, a comparison between population size estimates from Harris et al. (1995) and the current estimate is not advised.

Other evidence of changes through time

No other sources of data on temporal trends were found in the literature. However, a report from 1955 indicates a pre-breeding density of 118-530 field voles ha^{-1} (Lockie, 1955), and field vole 'plagues' have been regularly reported historically. Although population cycles continue, the pre-breeding densities for rough grassland in recent literature are at least half of those given in the 1955 report. This difference may reflect either a long-term change in population sizes, or simply that the available studies are not representative of the whole population. A summary of trends in population size and range is shown in Table 7.7d.

Table 7.7d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries		

Drivers of change

Table 7.7e Drivers of population change for field voles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Habitat fragmentation, nitrogen deposition and lack of management.		Negative

Data deficiencies

Table 7.7f Areas where further research is required to improve the reliability of population size estimates for field voles.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Bog Broadleaved woodland Urban and gardens Coniferous woodland Fen, marsh and swamp	Density estimates are based on expert opinion.
Multiannual population cycles.	All habitats	Adds uncertainty to the population size estimate as the stage of the cycle is unknown.
No occupancy data.	All habitats	
Availability of rough pasture of appropriate structure.	Grasslands	Field voles are highly dependent on tussocky grassland. Availability is currently difficult to determine from LCM2007.

Future prospects

Table 7.7g An assessment of the future prospects of the field vole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Decline

7.8 Orkney Vole *Microtus arvalis orcadensis*

Habitat preferences

The Orkney vole is a subspecies of the common vole *Microtus arvalis*. It has a larger body size than common voles, as well as other morphological differences (Berry, 1996). The species is present in most natural habitat types, conifer plantations and linear features throughout Orkney. However, it has largely disappeared from agricultural fields following the switch to high intensity production methods (Gorman and Reynolds, 1993; Gorman and Reynolds, 2003), and in these areas, ditches (including old peat-cuttings), fence-lines and road verges are important habitats.

Status

Non-native (naturalised — island endemic).

The species was introduced to the Orkney archipelago approximately 5,000 years ago (Martínková et al., 2013).

Conservation Status

- IUCN Red List (GB: VU; England: n/a; Scotland: [VU]; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Future prospects

Table 7.10h An assessment of the future prospects of the harvest mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline

7.11 Wood mouse *Apodemus sylvaticus*

Habitat preferences

The wood mouse is highly adaptable and is found in most habitats, including woodland, arable land, rough grassland, heather, blanket bog, sand dunes, urban areas and hedgerows (Kotzageorgis and Mason, 1997; Marsh and Harris, 2000; Flowerdew and Ellwood, 2001; Tattersall et al., 2001). Population densities in woodland vary with successional stage: mid-level regeneration of 5- to 10-year-old vegetation supports a higher density of wood mice than either ungrazed fields or 10-year-old regeneration (Marsh and Harris, 2000).

Hedgerows, including those distant from woodlands, are an important habitat for the wood mouse, and can support resident populations (Gelling et al., 2007). Population densities in hedgerows are high after arable crops are harvested, and also when grass swards are short because of grazing or cutting (Tew and Macdonald, 1993; Garratt et al., 2012). At these times, the wood mouse, like other small mammals, makes preferential use of boundary features rather than in-field areas (Tattersall et al., 2001). Similarly, hedgerows provide cover and food sources during autumn and winter (Kotzageorgis and Mason, 1997; Liu et al., 2013).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.11a. Gaps in the mainland species' distribution are likely to reflect a local lack of survey effort, rather than true absences. Further surveys are therefore recommended in these areas.

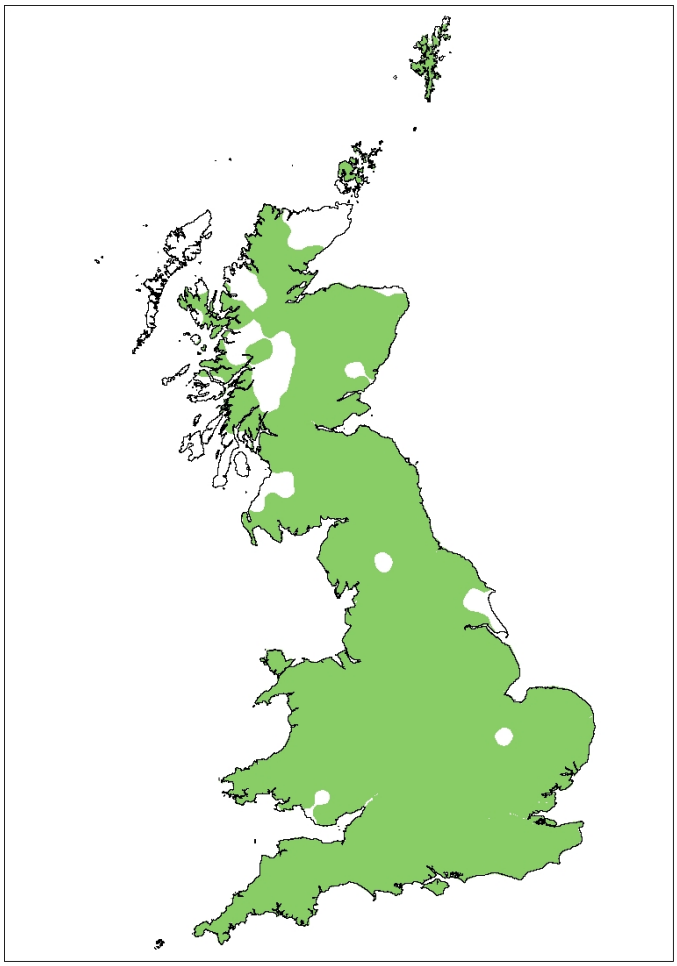


Figure 7.11a Current range of the wood mouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Wood mouse populations fluctuate seasonally, with peaks in autumn and early winter and troughs in spring and summer (Flowerdew, 1985; Harris et al., 1995). Density estimates were therefore derived only from studies which took place between March and June. As wood mice in arable land primarily use field margins, with incorporation of crop fields into home ranges only before the harvest (Tattersall et al., 2001), we assume that the in-field population will be included within the estimate for hedgerows. Arable land is therefore excluded from the analysis.

No recent density estimates were available for fen, marsh and swamp habitat. Had the estimates from Harris et al. (1995) for this habitat been included in the present review, confidence intervals would not have been calculable as none were provided in the original paper. In addition, the exclusion of fen, marsh and swamp altered the population estimate by <100,000 (<1%). This habitat class was therefore excluded.

Results

Eighteen relevant papers were identified by the literature search. Of these, five provided an estimate of pre-breeding population density, four provided a relative measure of abundance (captures per trap night), and one contained percentage occupancy for hedgerows (Gelling et al., 2007). The remainder provided post-breeding density estimates, explored the relationship of habitat variables to abundance but gave no effect size, or provided descriptive data only. Population density estimates are shown in Table 7.11a, and population size estimates in Table 7.11b.

Table 7.11a Median density estimates for wood mice, with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerows are divided into those managed or not through an agri-environment scheme (AES and non-AES). Hedgerow length and density per hedgerow are presented in km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	1,270,000	8.64	7.41	9.9	Attuquayefio et al. (1986) Flowerdew et al. (2004) Malo et al. (2012) Marsh and Harris (2000) Montgomery (1989)	1 162 1 19 158	n/a
Sand dunes	21,800	1.2	0.8	3.5	Attuquayefio et al. (1986) Gorman and Ahmad (1993)	1 8	n/a
Urban and gardens	1,360,000	4	0.05	8	expert opinion	2	n/a
Coniferous woodland	1,250,000	6.75	5.6	7.95	expert opinion	2	n/a
Dwarf shrub heath	1,540,000	1.5	0.1	3	expert opinion	1	n/a
Improved grassland	6,760,000	0.1	0	0.5	expert opinion	2	n/a
Unimproved grassland	1,060,000	6.5	0	13	expert opinion	2	n/a
Hedgerows (AES)	142,000 (km)	3	0	12	Broughton et al. (2014) Flowerdew et al. (2004)	11 9	93.9
Hedgerows (Non-AES)	320,000 (km)	14.62	12.31	24.5	Kotzageorgis and Mason (1997) Shore et al. (2005)	6 23	93.9

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.11b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 7.11a with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Area of suitable habitat (ha)*	Population size	-95%CI	+95%CI
England	7,260,000	22,700,000	11,600,000	37,800,000
Scotland	4,240,000	12,300,000	6,510,000	18,800,000
Wales	1,770,000	4,600,000	2,240,000	7,680,000
Britain	**13,300,000**	**39,600,000**	**20,400,000**	**64,300,000**

* The lengths of hedgerows are 393,000km in England, 17,600km in Scotland, and 51,000km in Wales.

Critique

No percentage occupancy data were available for most habitats; the population size is therefore overestimated. The population estimate is largely derived from broadleaved (28%) and coniferous (21%) woodland (Figure 7.11b), where population density estimates are supported by five (n=311) and four (n=49) references, respectively. Given the abundance of evidence relative to most other species in this review, sensitivity analyses were not conducted for these habitats.

Improved grassland forms 49% of the habitat within the geographical range of the wood mouse (Figure 7.11b), yet because of low population density ($0.1ha^{-1}$), it contributes just 2% of the estimated population. The density used for improved grassland is the median value from estimates provided by three expert opinions: given the extent of this habitat, field validation of the values would considerably improve the precision of the population estimate. A reliability assessment is shown in Table 7.11c. For density estimates based on expert opinion, a conservative score of 1 has been applied to the 'location of study sites' section.

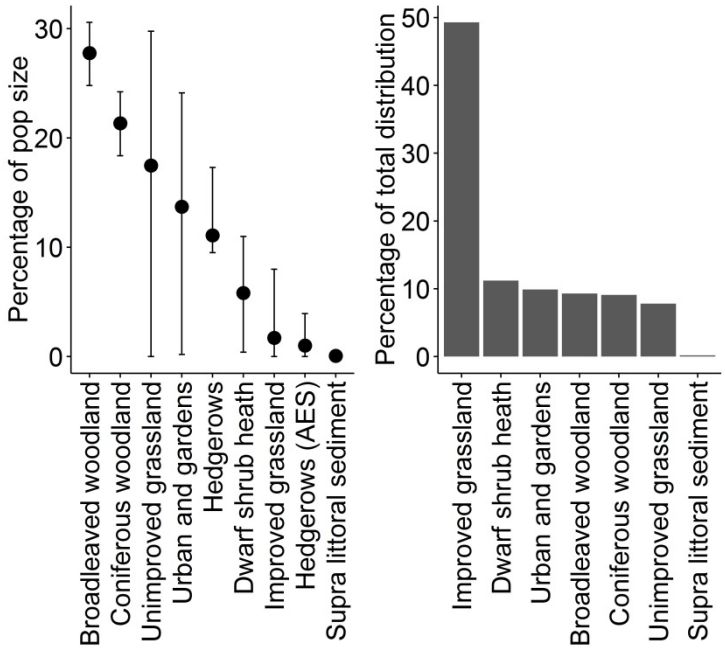


Figure 7.11b Left: The percentage of the total population of wood mice accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.11c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Broadleaved woodland	**Coniferous woodland**	**Improved grassland**
Location of study sites	0	Estimates from one location			
	1	Estimates restricted		1	1
	2	Estimates widespread	2		
Sample size	0	<10 density estimates		0	0
	1	10-30 density estimates			
	2	>30 density estimates	2		
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**4**	**1**	**1**
		Overall reliability score	**2**		

Changes through time

Comparison to Harris et al. (1995)

Population estimates from Harris et al. (1995) were derived from densities reported in the literature and by experts. The habitat classes were equivalent to those used in the current review. Population size was estimated as 38,000,000 in total, with 19,500,000 in England, 15,000,000 in Scotland and 3,500,000 in Wales. These figures are all within our confidence limits, so there is no evidence of a significant change in population size since 1995.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size that differs from the original estimate by only 5%, and that falls within the confidence limits of the original. It is therefore concluded that methodological differences have no material impact on the comparisons between the two time periods.

Other evidence of changes through time

No other references to a temporal trend in population size were found in the literature. A summary of trends in population size and range is provided in Table 7.11d.

Table 7.11d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable		All countries		
	Decrease				
	Data deficient				

Drivers of change

Table 7.11e Drivers of population change for wood mice between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Unknown			

Data deficiencies

Table 7.11f Areas where further research is required to improve the reliability of population size estimates for wood mice.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Urban and gardens Coniferous woodland Dwarf shrub heath Improved grassland Unimproved grassland	Estimates are based on expert opinion.
No occupancy data.	All habitats except hedgerows	
Density estimates are more than 10 years old.	Sand dunes Hedgerows	The most recent density estimates are from 1993 and 2005, respectively.
Limited density estimates are available for specified habitat.	Sand dunes	Nine individual density estimates are available.

Future prospects

Table 7.11g An assessment of the future prospects of the wood mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

7.12 Yellow-necked mouse *Apodemus flavicollis*

Habitat preferences

The yellow-necked mouse is found primarily in mature broadleaved woodlands (Marsh et al. 2008), particularly ancient coppiced woodlands, where they favour older established compartments, rather than recent coppice (Gurnell et al., 1992; Capizzi and Luiselli, 1996). Hedgerows also provide an important habitat for the species in Britain, with telemetry studies showing that individuals can reside solely within a linear hedgerow habitat (Montgomery, 1978). The availability and diversity of tree seeds is an important predictor of yellow-necked mouse density (Marsh et al., 2001). Britain is at the western edge of the species' European range, possibly owing to the impact of low summer temperatures on tree seed abundance. There is some potential for misidentification of this species with the wood mouse, particularly at the edges of its geographical range where abundance may be relatively low.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: n/a; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

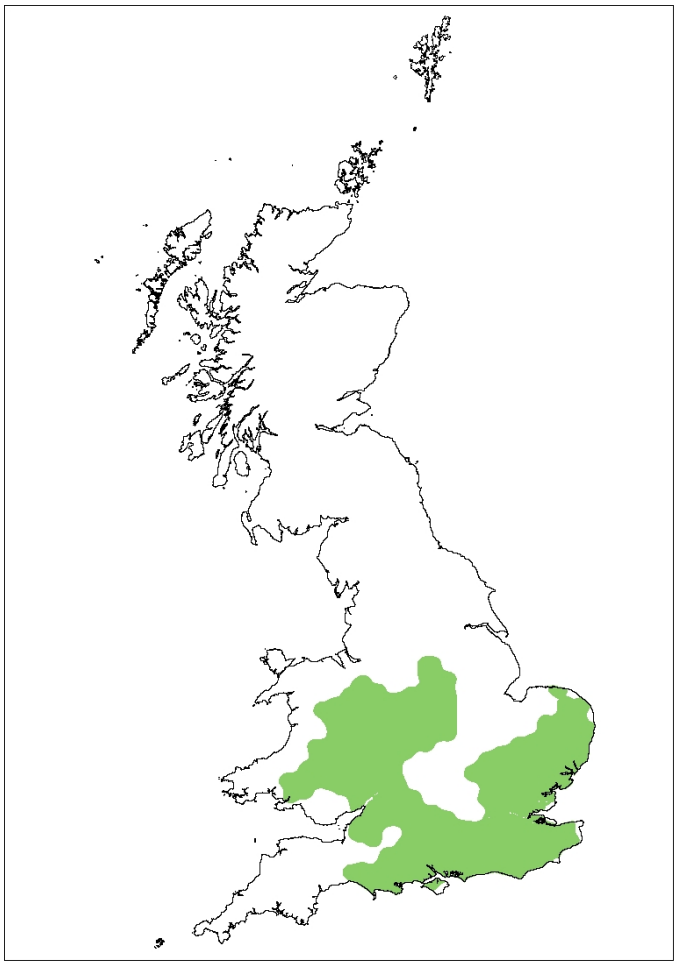


Figure 7.12a Current range of the yellow-necked mouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

The percentage of occupied sites for broadleaved woodland was taken from Marsh et al. (2001) where 80 of 146 sites (55%) within the species' range were occupied. Only non-zero population density estimates were included to avoid accounting for occupancy twice. The total area of broadleaved woodland within the species' distribution was multiplied by the percentage of occupied sites. The occupied area of broadleaved woodland was then used for all subsequent calculations of population size. In the absence of occupancy data for coniferous woodlands and urban areas, the same value (55%) was applied to these habitats. For hedgerows, percentage occupancy was taken from Gelling et al. (2007), where 180 hedgerows on 12 dairy farms in 4 geographical areas within the species' range were surveyed. Occupancy (75%) was provided for one of these areas, but in the other areas very

few yellow-necked mice were captured. To provide a more representative value for the species throughout its range, an average value of percentage occupancy was taken across all four areas, where the number of sites surveyed across all areas was assumed to be equal, and percentage occupancy at the remaining three sites was assumed to be roughly zero. As two of the sites were located towards the edge of the species' range where densities might be expected to be lower, the resulting value may be a slight underestimate.

Results

Six papers contained useful information for yellow-necked mice. Three of these reported pre-breeding population density (Montgomery, 1980; Kotzageorgis and Mason, 1997; Marsh and Harris, 2000), one gave occupancy for hedgerows (Gelling et al., 2007), and one gave occupancy for broadleaved woodland (Marsh et al., 2001). The remaining papers reported only post-breeding density estimates (Marsh et al., 2001; Moro and Gadal, 2007). Population density estimates are shown in Table 7.12a, and population size estimates in Table 7.12b.

Table 7.12a Median density estimates for yellow-necked mice with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Median population density from Marsh et al. (2001) was taken from positive sites only. Hedgerow length and density per hedgerow are presented in km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (per ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	592,000	3.12	0.57	11	Montgomery (1980) Marsh and Harris (2000)	12 13	55%
Urban and gardens	720,000	0.38	0.12	0.8	expert opinion	2	55%
Coniferous woodland	160,000	0.16	0.08	0.52	expert opinion	2	55%
Hedgerows	200,000 (km)	8.5	6	11	Kotzageorgis and Mason (1997)	6	19%

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.12b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 4.1a with the area of habitat within the species' distribution, and adjusting for occupancy.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	1,330,000	1,360,000	426,000	3,940,000
Wales	146,000	140,000	40,600	423,000
Britain	**1,470,000**	**1,500,000**	**467,000**	**4,360,000**

* The lengths of hedgerows are 185,000km in England and 15,300km in Wales.

Critique

68% of the population estimate for the yellow-necked mouse is derived from broadleaved woodland, a habitat which forms 40% of the species' range (Figure 4.1a). The density values

applied for broadleaved woodland are based on 25 separate estimates from two papers, although the confidence limits are wide (0.57-11ha^{-1}), reflecting highly variable densities across the geographical range. Further surveys, specifically designed to include a representative sample, would improve confidence in the estimate.

Hedgerows contribute 21% of the estimated population, and there are 200,000km available within the species' range (Table 7.12a, Figure 7.12b). The abundance estimate for hedgerows is derived from 6 estimates from one paper, with the resulting median density estimate being substantially higher than that in broadleaved woodland. Re-calculation of population size following stepwise removal and replacement of the individual density estimates for hedgerows did not result in a significant change in population size. Reliability scores are shown in Table 7.12c. For the purposes of this assessment, we consider the population density estimates of experts to be representative of a restricted area of the species' range.

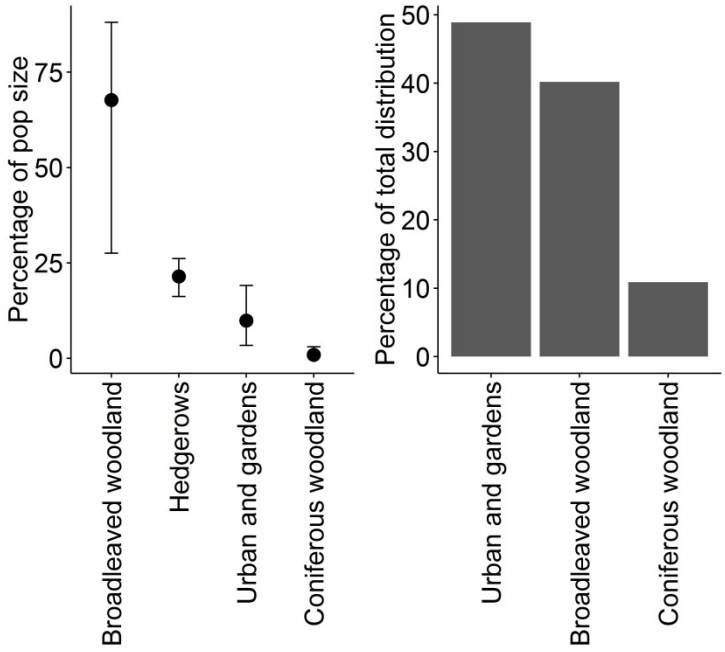


Figure 7.12b Left: The percentage of the total population of the yellow-necked mouse accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.12c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Broadleaved woodland	**Urban and gardens**
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates		0
	1	10-30 density estimates	1	
	2	>30 density estimates		
Occupancy data available?	0	No		
	1	Yes	1	1
		Habitat score	**3**	**2**
		Overall reliability score	**2.5**	

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated the total population size for Britain as 750,000, with 662,500 in England and 87,500 in Wales. The British estimate was based on an estimate of 450,000 for ancient woodlands, with the remaining 300,000 added to allow for yellow-necked mice in other habitats. The current review uses broadleaved woodland (adjusted for 55% occupancy), rather than ancient woodland.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size that differs from the original estimate by only 5%, and that falls within the confidence limits of the original. It is therefore concluded that methodological differences have no material impact on the comparisons between the two time periods.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is provided in Table 7.12d.

Table 7.12d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	England* Wales			

* Limited expansion into the Midlands.

Drivers of change

Table 7.12e Drivers of population change for yellow-necked mice between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Decline in habitat quality.	Change in the management of ancient/coppiced woodlands, although evidence for the effect on yellow-necked mice is sparse.	Harris et al. (1995)	Negative
Climate change, causing a rise in summer temperature.	Range expansion: the availability of food (tree seeds) is linked to high summer temperature.	Marsh et al. (2001)	Positive

Data deficiencies

Table 7.12f Areas where further research is required to improve the reliability of population size estimates for yellow-necked mice.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Urban areas and gardens Coniferous woodland	Density estimates are based on expert opinion only.
Limited density estimates for key habitat.	Hedgerows	Fewer than 10 density estimates available.
Density estimates are more than 10 years old, highly variable, and possibly not representative of the species' geographical range.	Broadleaved woodlands Hedgerows	Estimates are from 2000.

Future prospects

Table 7.12g An assessment of the future prospects of the yellow-necked mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Potential increase
Range	Potential increase
Habitat	Stable

7.13 House mouse *Mus musculus*

Habitat preferences

The house mouse lives commensally with humans: its movement patterns and current widespread distribution are attributed to this relationship (Searle et al., 2009). Although listed as native to Britain by IUCN, the best available evidence suggests the species arrived in western Europe in the Bronze age and is recorded in Britain by the Iron age (Yalden, 1999). The species is therefore not subject to many of the environmental factors that regulate the population sizes of most other small mammals. It is, however, sensitive to human activities, including the alteration of buildings and the deployment of rodenticide (Pocock et al., 2004). The decline in urban infestations in the 1970s is likely to be the result of increased rodenticide efficacy (Richards, 1989; Harris et al., 1995). Detailed analyses of the habitat preferences of the house mouse are, however, lacking.

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: LC; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Future prospects

Following a feasibility study for the reinforcing of small populations of pine martens in England and Wales through translocation of individuals from Scotland, habitat suitability was assessed, and five regions in Wales and one on the English/Welsh border were identified as potential release sites (MacPherson et al., 2014). Thirty-nine pine martens were translocated successfully to central Wales during 2015 and 2016. The locations selected for release offered large areas of suitable habitat, and had low risks of mortality from road traffic incidents (MacPherson et al., 2014).The prospects for achieving an established population in Wales are therefore positive.

Table 8.5i An assessment of the future prospects for the pine marten, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

8.6 Stoat *Mustela erminea*

Habitat preferences

The stoat is found in most habitats in Britain, and at any altitude, provided that sufficient cover and prey are available. Like other mustelids, there is territory defence against intruders of the same sex (Erlinge, 1977), and the smaller home ranges of females tend to be overlapped by one or more males (Powell, 1979). The species tends to avoid open spaces by travelling along hedgerows, ditches and stone walls. The stoat is a specialised predator of small and medium-sized mammals, and rabbits are a key prey item, particularly for males, forming more than 50% of their diet throughout the year (McDonald et al., 2000). Foraging is therefore concentrated on rabbit warrens, early successional communities favoured by field voles, and brush timber piles that might harbour small mammals (see Harris and Yalden, 2008). Populations appear to fluctuate in response to food supply. They showed marked

declines (as measured by the National Gamebag Census) in the 1950s and 1960s, following myxomatosis epidemics (Sumption and Flowerdew, 1985).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [NT]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

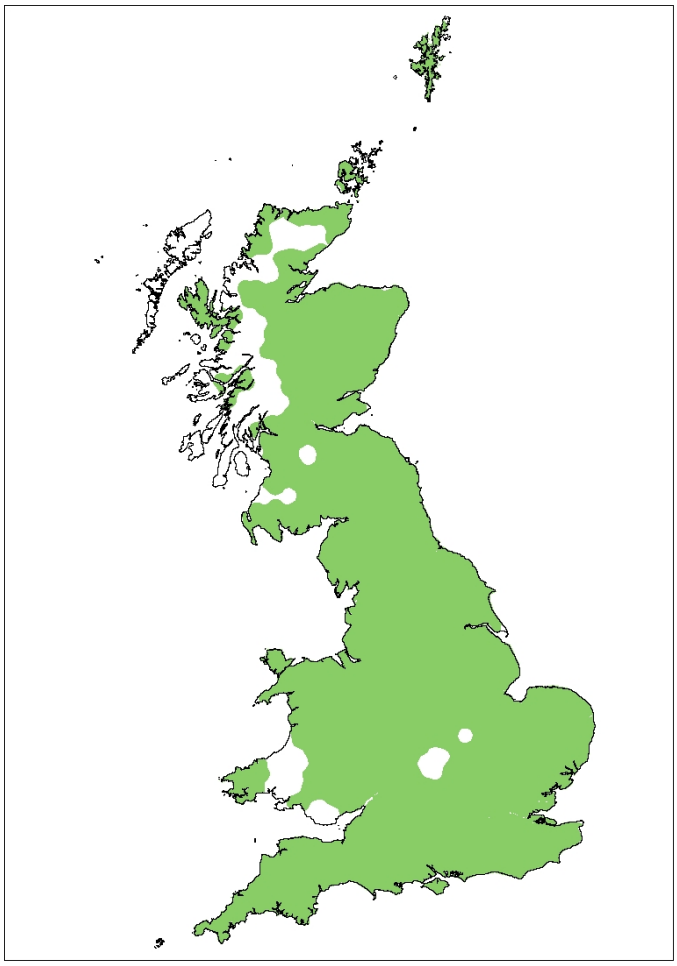


Figure 8.6a Current range of the stoat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

No papers identified by the literature search for stoats reported pre-breeding population density estimates, trends, occupancy, or the effect of environmental variables on relative density. The population density estimates (Table 8.6a) are therefore taken from Harris et al. (1995), but these were not based on any published data for Britain. Population size estimates are shown in Table 8.6b.

Table 8.6a Median density estimates with 95% confidence intervals for stoats, calculated using data obtained from Harris et al. (1995).

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Arable and horticulture	61,200	2	-	-	Harris et al. (1995)	1	n/a
Bog	6,700	2	-	-	Harris et al. (1995)	1	n/a
Broadleaved woodland	12,300	6	-	-	Harris et al. (1995)	1	n/a
Coniferous woodland	11,800	6	-	-	Harris et al. (1995)	1	n/a
Dwarf shrub heath	15,100	2	-	-	Harris et al. (1995)	1	n/a
Improved grassland	65,200	1	-	-	Harris et al. (1995)	1	n/a
Unimproved grassland	10,300	6	-	-	Harris et al. (1995)	1	n/a
Sand dunes	200	2	-	-	Harris et al. (1995)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.6b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.5a with the area of habitat within the species' distribution. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	116,000	[260,000]	-	-
Scotland	51,600	[140,000]	-	-
Wales	15,600	[37,600]	-	-
Britain	**183,000**	**[438,000]**	**-**	**-**

Critique

No percentage occupancy data were available; the population size is therefore overestimated. The habitat contributing the greatest proportion of the population estimate is arable land (28%), with a further 62% split between broadleaved woodland, coniferous woodland, improved and unimproved grassland (Figure 8.6b). Two of these habitats — improved grassland and arable land — contribute a high proportion of the population size because of the large areas present within the species' distribution (69%). In contrast, the high population densities in the other three habitats — unimproved grassland, coniferous and broadleaved woodland — explain their contribution to the population estimate (Table 8.6b).

Although stoats are present in arable land, they are more likely to use field boundaries and hedgerows. Home ranges in the Swiss Jura mountains tend to be linear and follow boundary features (Debrot and Mermod, 1983). Density estimates from Harris et al. (1995) were based on the authors' expert opinion, rather than empirical data, and it is unclear how this behaviour was taken into account for the density estimates for arable land. In this review, lengths of hedgerows were not included as a separate habitat category in order to avoid double counting.

To assess reliability, we have considered the population density estimates from Harris et al. (1995) to be the expert opinion of the authors and not representative of the entire species' range (Table 8.6c).

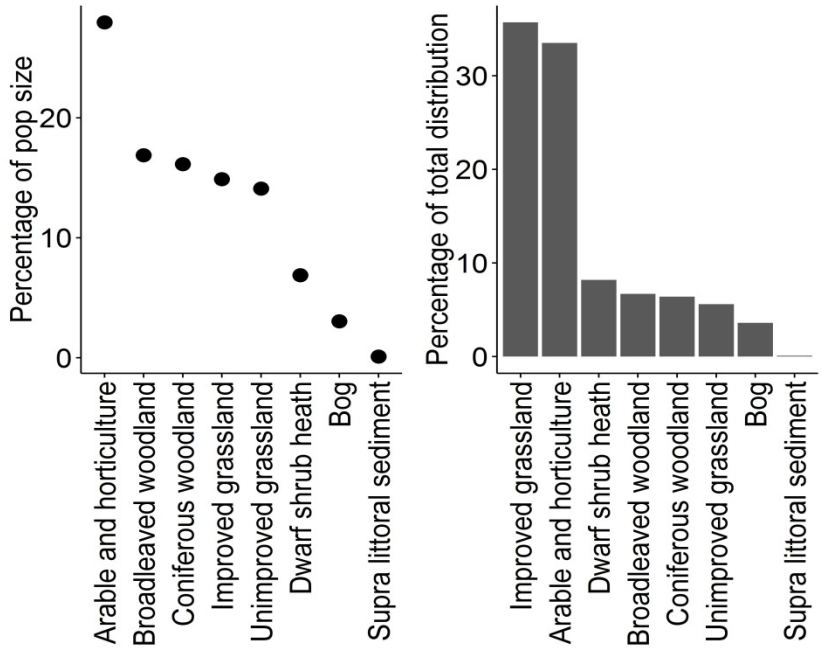


Figure 8.6b Left: The percentage of the total population of stoats accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 8.6c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Arable and horticulture	Improved grassland
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 population density estimates	0	0
	1	10-30 population density estimates		
	2	>30 population density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**1**
		Overall reliability score	**1**	

Change through time

Comparison to Harris et al. (1995) and Arnold (1993)

Total population size as estimated by Harris et al. (1995) was 462,000, comprised of 245,000 in England, 180,000 in Scotland and 37,000 in Wales. Population density estimates for all habitat types are based on the authors' expert opinion (Harris et al., 1995). The density estimates in the current review are taken from Harris et al. (1995), so any differences in population size would be the result of divergent measurements of habitat availability.

Nationally, there are changes between the two reviews in the estimated availability of key habitats, generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a new population size estimate of 399,000, which is 9% smaller than the estimate shown in Table 8.6b. The significance of this reduction cannot be assessed because of the lack of confidence limits for either estimate. The original estimate is close to the value presented by Harris et al. (1995), whereas the adjusted estimate is 14% lower.

Other evidence of changes through time

The National Gamebag Census suggests an increase of 28% (95% CI = 12%-42%) in the numbers of stoats culled between 1995 and 2009 (Aebischer et al., 2011), although it is unclear whether this increase is owing to an actual population size increase or an increase in trapping effort. A summary of trends in population size and range is provided inTable 8.6d.

Table 8.6d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries*		

*Aebischer et al. (2011).

Drivers of change

Table 8.6e Drivers of population change for stoats between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Change in prey availability.	A decline in specialist prey species, although the significance is unknown.	Harris and Yalden (2008)	Negative

Data deficiencies

Table 8.6f Areas where further research is required to improve the reliability of population size estimates for stoats.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	All habitats	All density estimates are taken from Harris et al. (1995), which were based the authors' opinions.
Density estimates do not represent within-habitat variability.	All habitats	No range or confidence limits were available from Harris et al. (1995).
Managed populations.	All habitats	The population size has not been adjusted to reflect the number culled.
No occupancy data.	All habitats	

Future prospects

Table 8.6g An assessment of the future prospects for the stoat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Stable

8.7 Weasel *Mustela nivalis*

Habitat preferences

The weasel occupies a wide range of habitats. As with other mustelids, there is territory defence against intruders of the same sex, and the smaller home ranges of females tend to be overlapped by one or more males (Powell, 1979). Home range size, and hence density, is strongly dependent on food availability. The weasel is a specialised predator of voles and mice, but will also take young rabbits, birds, and birds' eggs, particularly in spring when rodent populations are low (McDonald et al., 2000). Common in coniferous woodlands with dense field vole populations, the species is less abundant where small mammals are scarce, such as at high altitudes or in deciduous woodlands with sparse ground cover. On farmland, it is strongly associated with hedgerows, stone walls and other linear features, and rarely ventures into open habitat (see Harris and Yalden, 2008).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

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Research

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An experimental investigation into the effects of traffic noise on distributions of birds: avoiding the phantom road

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Many authors have suggested that the negative effects of roads on animals are largely owing to traffic noise. Although suggestive, most past studies of the effects of road noise on wildlife were conducted in the presence of the other confounding effects of roads, such as visual disturbance, collisions and chemical pollution among others. We present, to our knowledge, the first study to experimentally apply traffic noise to a roadless area at a landscape scale—thus avoiding the other confounding aspects of roads present in past studies. We replicated the sound of a roadway at intervals—alternating 4 days of noise on with 4 days off—during the autumn migratory period using a 0.5 km array of speakers within an established stopover site in southern Idaho. We conducted daily bird surveys along our 'Phantom Road' and in a nearby control site. We document over a one-quarter decline in bird abundance and almost complete avoidance by some species between noise-on and noise-off periods along the phantom road and no such effects at control sites—suggesting that traffic noise is a major driver of effects of roads on populations of animals.

1. Introduction

Roads are prevalent across vast stretches of the Earth and 83% of the USA is within 1 km of a road [1]. Although some studies have shown positive effects of roads on wildlife, the cumulative effects across taxa are overwhelmingly negative (reviewed by [2]). A recent meta-analysis of 49 datasets including 234 species of mammals and birds across four continents demonstrated that bird and mammal populations decline within 1 and 5 km of human infrastructure—including roads—respectively [3]. Despite myriad studies regarding the effects of roads on wildlife, the primary mechanism underlying these effects remains unknown. Effects of roads include habitat fragmentation, road mortality, sensory disturbance and chemical pollution, among others (reviewed by [2]). Therefore, a given road probably impacts wildlife in several ways—making it exceedingly difficult to estimate the strength of any single effect.

Several lines of evidence suggest that traffic noise is a major factor explaining declines in populations of wildlife near roads. Perhaps the best tests of the effects of noise on animal distributions come from studies of noise produced by natural gas compressors. For example, Bayne *et al.* [4] found a one-third reduction in songbird density at noisy gas compressor stations in Canada compared with nearby well pads that were almost identical in habitat, but were much quieter. Francis *et al.* [5] used a similar system of noisy gas compressors and quiet well pads in New Mexico to show that species richness of birds is greatly reduced at noisy sites, with 14 species avoiding areas surrounding gas compressors. Evidence from gas fields indicates that species remaining in noisy areas are those that vocalize within frequencies less masked by anthropogenic noise [6,7]—a phenomenon probably occurring in areas exposed to traffic noise. For example, birds that vocalize at frequencies similar to those of road noise are more likely

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to avoid roads than species that vocalize at higher frequencies [8,9]. Species of frogs, birds and mammals are known to change the characteristics of their vocalizations in the presence of traffic and other anthropogenic noise, presumably to avoid the effects of masking (reviewed by [10]).

Behavioural evidence and studies of other anthropogenic noise sources suggest that road noise should be a major driver of observed road effects. However, although road ecology studies attempting to directly address the effect of traffic noise on wildlife have suggested that noise is a major cause of negative effects, they are typically conducted in the presence of other effects of roads [11]. For instance, Halfwerk *et al*. [12] demonstrated that great tits (*Parus major*) have reduced reproductive success in areas exposed to high levels of road noise. Several studies from The Netherlands have shown that bird distributions near roads are negatively associated with noise levels (reviewed by [2]). A study in the USA [13] found that distributions of grassland birds were negatively associated with traffic volume (vehicles per day)—an effect that was attributed to increased noise levels. However, Summers *et al*. [11] found that the effect of distance to a road was stronger than the effect of noise level and suggested that road mortality, not noise, was probably underlying the negative effects of roads. As Summers *et al*. [11] suggest, these past studies attempting to assess the effects of road noise on wildlife are certainly informative, but are confounded by other effects which are present at any road. Put another way, the current study paradigm in road ecology of comparing roadless areas to sites near roads or using observational data to examine correlations between road noise and animal abundance is perhaps yielding diminishing returns.

Noise playback is an effective method of testing the effects of noise in the absence of other factors but has been underused in road ecology because of the difficulty of applying noise across a landscape [14]. Recently, however, experimental application of noise has become more common in studies of road ecology. For example, Crino *et al*. [15] used playback to examine the effects of road noise on stress in nestling white-crowned sparrows (*Zonotrichia leucophrys*). Arroyo-Solís *et al*. [16] experimentally demonstrated that the spotless starling (*Sturnus unicolor*) and the house sparrow (*Passer domesticus*) shift the timing of their morning songs in response to urban noise. Blickley *et al*. [14] used experimental playback to test the effects of noise on the distribution of the greater sage-grouse (*Centrocercus urophasianus*)—demonstrating reduced lek attendance at sites experimentally exposed to road and gas-compressor noise. Here, we assess the effects of road noise in the absence of the other effects of roads by adding road noise to a roadless landscape—allowing us to experimentally test the effects of road noise on an entire community of migrating birds. We applied noise using a 0.5 km array of speakers—which we term the 'phantom road'—situated at an autumn migratory stopover site in the Boise Foothills in south-western Idaho. We use a modified before–after-control-impact experimental design to assess changes in the abundance of migrating birds near the phantom road with the speakers turned on and off in repeating 4 day intervals in relation to a nearby control site.

We chose to apply our study design to migrating birds for two reasons. First, because migrating birds stay at stopover sites for short durations, the bird community is constantly changing. This constant change in the bird community means that as we turn the phantom road on and off over the course of the migratory period we sample different individuals and populations of birds. Second, populations of migratory birds have declined sharply in recent decades [17–20] and identification, preservation and maintenance of stopover habitat is a research priority [21–24]. To our knowledge, the effects of anthropogenic noise on birds during migration have never been examined. Accordingly, our study fills an important gap in our knowledge of the use of stopover habitat. We therefore test the hypotheses that animals avoid roads because of disturbance by noise and that anthropogenic noise degrades migratory stopover habitat. We predicted that fewer birds would be present near the phantom road when speakers were turned on than when they were off, and that bird abundance would be negatively correlated with sound levels.

2. Material and methods

(a) Phantom road

We constructed the phantom road on an east–west oriented ridge extending southeast from Lucky Peak, Idaho roughly 0.8 km from the Idaho Bird Observatory's field site. This ridge was typical of most ridges along the Boise front in that the vegetation on north-facing slopes was dominated by mature Douglas fir forest (*Pseudotsuga menziesii*), whereas on the south-facing slopes bitter and choke cherry bushes (*Prunus virginiana* and *emarginata*) were prevalent on drainages and sagebrush steppe dominated ridges. We erected 15 pairs of speakers in Douglas fir trees along the crest of the ridge—with one speaker oriented towards the evergreen forest, and the other oriented towards the cherry/sage. Each set of speakers was approximately 4 m above the crest of the ridge. We amplified the speakers (Dayton Audio—Springboro, OH, USA—RPH16 Round 16′ PA Horns paired with MCM Electronics—Centerville, OH, USA—40 W midrange compression drivers ($\pm$5 dB(A), 400–3000 Hz)) with Parts Express (Springboro, OH, USA) 2 W x 2channel, 4-ohm, Class D amplifiers and played back sound files (MP3, 128 kbps) using Olympus (Center Valley, PA, USA) LS-7 and Roland (Los Angeles, CA, USA) R-05 audio players. We powered amplifiers and audio players with arrays of $LiFePO^4$ (Batteryspace, CA, USA) batteries housed in waterproof plastic containers. The geometry of a sound source can have profound impacts on the scale of noise exposure—point sources (e.g. generators, gas-compressor stations, a single car) lose sound energy at approximately 6 dB per doubling of distances, whereas line sources (e.g. a busy roadway, train) fall off at approximately 3 dB per doubling of distance. We therefore placed speakers roughly 30 m apart to ensure that, when playing noise, the individual speaker point sources fused into a line-source at our bird count locations. The phantom road was therefore roughly 0.5 km in length (figure 1*a*).

We applied noise along the phantom road at 4 day intervals—alternating between noise-on and noise-off periods throughout autumn migration. We alternated every 4 days because almost all species stopover at our site for fewer than 8 days on average during autumn migration [25]. Therefore, individual birds were probably only present during one noise-on and one noise-off period, and thus each set of noise-on/off periods was probably independent. Alternating between noise-on and noise-off periods also ensured that noise-on and noise-off periods were not correlated with potentially confounding factors, such as seasonally variable phenology of migrating birds, fruit or insect availability and weather fronts. We attempted to mimic normal traffic patterns by beginning noise playback at 04.30 and ending it at 21.00 local time each day during noise-on blocks. We gradually increased the volume of the noise over

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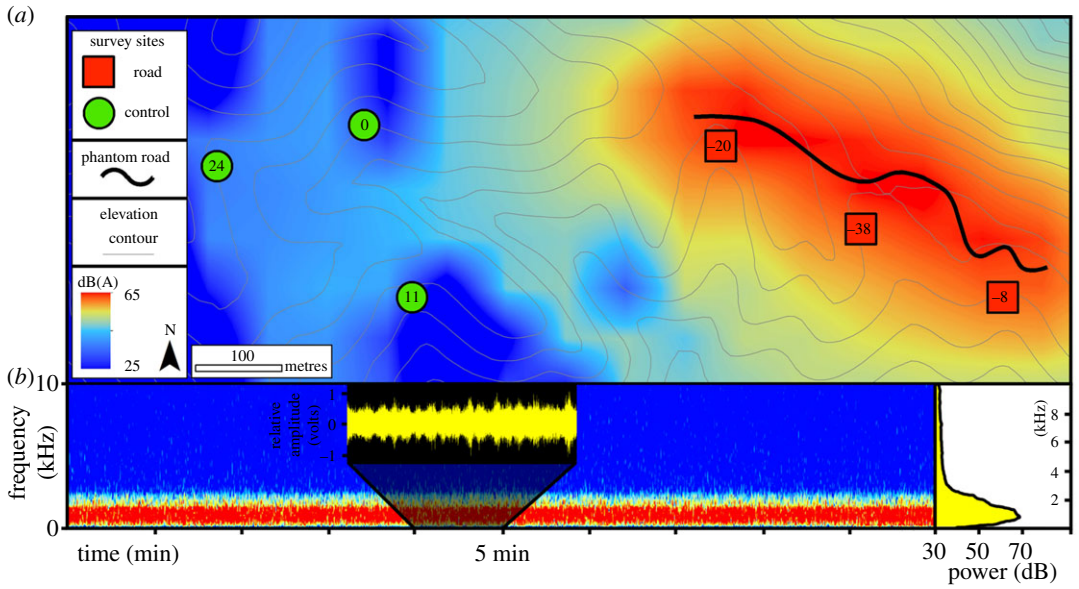


Figure 1. (*a*) Estimated background sound levels (dB(A) 1 h LEQ) during periods when speakers were turned on at our study site in the Boise Foothills in south-western Idaho. Background sound level was modelled using NMSIM (Noise Model Simulation; Wyle Laboratories, Inc., Arlington, VA) where inputs were chosen to match observed values at point count locations. Numbers represent the per cent change in birds present at a survey site between 4 day noise-on and noise-off periods which alternated continuously from 19 August through to 9 October 2012. (*b*) Ten minute spectrogram, 1 min waveform and power spectrum of noise file (recorded 35 m from the phantom road at a point count location) played from 04.30 to 21.00 along the phantom road during noise-on periods.

30 min in the morning so as not to elicit a startle response from any birds along the phantom road and also gradually decreased the volume over 15 min at the end of each noise-on day.

(b) Playback files

We played traffic noise recorded within Glacier National Park. To create the playback file, we combined files of 12 individual cars recorded at known distances, decibel levels and speeds. We choose car pass-by events based on clarity of recording, decibel level and speed. We created a 1 min file of 12 car pass-by events and repeated this file without shuffling. Because any possible habituation would have only reduced our ability to detect distributional changes, we see this as a minor concern. Our playback file therefore contained 720 pass-by events per hour of cars travelling at approximately 45 miles h^{-1}—traffic levels and speeds found along roads in some of the most visited national parks, national forests and other protected areas globally. Our playback file further simulated the frequency profile of typical traffic noise with most of the energy of the noise between 0 and 3 kHz with a peak around 1 kHz (figure 1*b*). We manipulated the source level of each set of speakers to produce a background decibel level of roughly 55–60 dB(A) hourly level-equivalent (LEQ) at each of the three point count locations along the phantom road. Hourly LEQ values are the level of a constant sound over an hour that has the same energy of the actual, fluctuating energy over that hour [26]. The background noise level during noise-on periods therefore approximated the thresholds hypothesized by Reijnen *et al.* [27] at which road noise negatively affects densities of birds (also see [26]).

(c) Bird counts

We placed point count locations along the phantom road within the centres of three patches of cherry shrubs. We surveyed patches of cherry shrubs because they contain the highest diversity and abundance of birds of any habitat-type within our study site during migration [28]—allowing us to maximize sample size while examining the effects of noise on the high-quality stopover habitat. Each point count location was at least 150 m from the nearest other location, and between 30 and 50 m from the nearest speaker. Because patches of cherries were within drainages, each point count location was also separated from the other locations by ridges, meaning that an observer was unlikely to detect birds in patches of cherries other than the one they were currently surveying. We also placed control point count locations at three locations roughly 0.8 km away from the phantom road, also within patches of cherries separated by ridges (figure 1). Control point count locations were placed on sites with roughly the same slope and aspect and at roughly the same distance to forest as sites along the phantom road and all point count locations were within 100 m in elevation of each other. We therefore made every effort to ensure that control and experimental sites were placed within the same habitat mosaic and within similar microhabitats.

We conducted point counts at all point count locations along the phantom road and at the control site daily within 5 h after sunrise following a modified protocol of the Rocky Mountain Bird Observatory (D. J. Hanni, C. M. White, R. A. Sparks, J. A. Blakesley, G. J. Levandoski, J. J. Birek 2009, unpublished report). At each point count location, observers conducted three consecutive 5 min point counts [29]. For each individual bird detected during a point count, observers recorded the species and the minute in which it was first detected, as well as the method of detection (visual, call or song). Observers also recorded the distance to a bird when it was first detected using a laser range-finder. To increase our ability to obtain an accurate distance measurement, observers performed point counts in 4 m tall towers which allowed them to more easily detect a bird above the dense shrub layer. Observers also recorded temperature, wind speed (Beaufort Scale) and per cent cloud cover upon arriving at each survey location. Because detection of birds varies by both time and date, we shuffled the order in which points were surveyed every day. We alternated which site (control or phantom road) was surveyed first every 8 days, coinciding with the changes in noise-on/off blocks. Furthermore, the order of point count locations surveyed within each site was randomly determined for each day.

Two trained observers conducted point counts during our study, with one observer conducting all counts on a given day. Because probability of detection is probably different between observers, we scheduled observers so that they surveyed the

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same number of days during the season ($n = 26$) and both observers conducted at least one survey during each noise-on and noise-off block. Our sampling scheme therefore ensured that all locations were surveyed at different times of the morning by different observers throughout the study season.

Because background noise levels exceeding 45 dB have been shown to negatively affect the probability of detection of birds within 60 m [30], observers turned off the speakers surrounding individual point count locations before counts at sites near the phantom road during noise-on blocks. Observers only turned off speakers surrounding the location that they were currently surveying and turned them back on before moving to the next location. Turning off speakers in this manner ensured that dB levels were below 45 dB (confirmed by a dB metre) during point counts, and minimized the time speakers needed to be turned off during noise-on mornings.

(d) Analysis of background sound level

We determined the background sound levels of each point count location during noise-on and noise-off blocks using MP3 audio recordings [31]. During two full noise-on and noise-off blocks, we deployed an MP3 recorder (Roland R05 or R09 recording at 128 kbps) at each point count location, which continuously recorded background sound level during the entire blocks. We then used a custom programme (Damon Joyce, NPS, AUDIO2NVSPL) to convert the MP3 recordings into an hourly sound pressure level format, and then converted those values to hourly LEQ values in dB(A) using another custom programme (Damon Joyce, NPS, Acoustic Monitoring Toolbox). Finally, we averaged the hourly background LEQ during the hours of 05.00 through to 21.00 across the noise-on and noise-off blocks to create the noise-on and noise-off LEQs.

(e) Statistical analysis

To ensure that we only examined birds within the patches of cherries that we intended to survey, we truncated the data to include only birds detected within 50 m of the sample point. Although our sampling scheme was designed to minimize the effects of heterogeneity in probability of detection, we evaluated and corrected our counts for the possibility of imperfect detection using a removal model [32]. A removal model calculates the probability of detecting a present individual during a survey using the minute in which individual birds are detected during surveys. We implemented the removal model using the Huggins closed-capture setting in MARK [33] using the package RMark [34] in R [35]. We built models of detection including combinations of observer, noise-on versus noise-off, control versus experimental sites, and linear and quadratic effects of date. We also built a null model which only included the intercept and a global model that included all factors. We ranked and compared the models using Akaike's Information Criterion (AIC, [36]) corrected for small sample size (AICc, [37]). We then used the estimates from the highest ranked detection model to calculate the probability of detecting an individual if it was present during each survey and corrected the observed count of each survey for detection by dividing the observed count by the probability of detection during that survey [32]. Although inference from this study with and without a correction for detection is qualitatively similar, we present results of the detection-corrected analysis.

Once our counts were corrected for imperfect detection, we modelled the abundance of birds at our survey locations in response to site and seasonal differences as well as changes in background noise levels owing to the phantom road. We modelled abundance using linear mixed-effects models and controlled for the repeated sampling of sites using a random intercept for each point count location. We also controlled for possible temporal autocorrelation by including an autoregressive error structure within each model. Furthermore, plots of bird abundance against background dB(A) levels revealed possible problems regarding heteroskedasticity among residuals. For each species, we therefore tested a null model which included a 'power of the covariate' error structure with dB(A) as the covariate against one with the default error structure. We then ranked the models with competing error structures using AIC and used the error structure within the AIC-best model within all subsequent models for that species. We also normalized abundance values before analysis using a natural log transformation.

We built linear mixed-effects models representing several hypotheses regarding bird abundance. Each model included a random intercept for survey site. We constructed models representing several *a priori* hypotheses (see the electronic supplementary material, appendix S1). Several models included an interaction between factors indicating whether an observation occurred at the control site or along the phantom road and a factor representing noise-on and noise-off periods as well as their main effects. This interaction model represents the hypothesis that abundance along the phantom road changed between noise-on and noise-off periods while there was no difference in abundance between noise treatments at the control site. The main effect of site controls for potential differences in habitat between the control site and the phantom road. Several models contained the dB(A) levels recorded at each site during noise-on and noise-off periods—representing the hypothesis that bird abundance is linearly related to background sound levels. We also controlled for seasonal fluctuations in bird abundance by building models that contained linear and quadratic effects of Julian date. All mixed models were built in R using the package nlme [38] and were fit using maximum likelihood. Because models within $\Delta AIC < 2$ are considered to receive equal support as the best models [39], we considered there to be an effect of noise on bird abundance if models within $\Delta AIC < 2$ contained either an interaction between site and noise factors, or the covariate for background dB(A) levels with 85% CIs of these terms excluding zero [40]. Because of convergence and over-fitting problems inherent with small sample sizes, we only analysed data for species with more than 50 detections.

3. Results

We recorded 8078 detections of birds of 59 species within 50 m of bird survey locations (table 1). Twenty-two species were detected more than 50 times and those species constituted 91% of the total detections within 50 m of the observer (table 1). We integrated a total of 120 h of background noise levels for noise-on and noise-off blocks, separately at each survey site. The noise-on LEQ at point count locations near the phantom road averaged 55 (s.e. = 0.6) dB(A) and was 11 (s.e. = 2.6) dB(A) greater than the average noise-off LEQ along the phantom road. Whereas, noise-on LEQ at control locations averaged 41 (s.e. = 1.8) dB(A) and was 1 (s.e. = 0.2) dB(A) greater than the noise-off LEQ. The range of hourly LEQ values (L_{min}–L_{max}) during noise-on periods were 31 (s.e. = 4.8)–51 (s.e. = 0.7) at control survey locations and 36 (s.e. = 2.5)–63 (s.e. = 1.3) at road survey locations. Whereas during noise-off periods hourly LEQ values ranged from 31 (s.e. = 4.8) to 45 (s.e. = 0.2) at control survey locations and 32 (s.e. = 6.1)–52 (s.e. = 2.5) at road survey locations. The slightly higher background sound level during noise-off periods near the phantom road compared with control sites was probably owing to wind exposure. Overall, our study design produced a gradient of sound levels ranging from roughly 37 to 57 dB(A) under which birds were sampled.

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Table 1. Common name, scientific name and number of encounters of birds detected within 50 m of point count locations within the Boise Foothills of southern Idaho 19 August through 9 October 2012. (Unidentified birds are not listed.)

common name	scientific name	no. encounters
American robin	*Turdus migratorius*	1452
ruby-crowned kinglet	*Regulus calendula*	890
spotted towhee	*Pipilo maculatus*	877
dark-eyed junco	*Junco hyemalis*	600
white-crowned sparrow	*Zonotrichia leucophrys*	583
yellow-rumped warbler	*Setophaga coronata*	564
red-breasted nuthatch	*Sitta canadensis*	560
Cassin's finch	*Haemorhous cassinii*	274
Cassin's vireo	*Vireo cassinii*	193
cedar waxwing	*Bombycilla cedrorum*	181
pine siskin	*Spinus pinus*	173
western tanager	*Piranga ludoviciana*	148
chipping sparrow	*Spizella passerina*	143
mountain chickadee	*Poecile gambeli*	129
MacGillivray's warbler	*Geothlypis tolmiei*	119
Townsend's solitaire	*Myadestes townsendi*	81
yellow warbler	*Setophaga petechia*	76
lazuli bunting	*Passerina amoena*	64
golden-crowned kinglet	*Regulus satrapa*	62
dusky flycatcher	*Empidonax oberholseri*	59
Townsend's warbler	*Setophaga townsendi*	52
evening grosbeak	*Coccothraustes vespertinus*	51
brown creeper	*Certhia americana*	49
Wilson's warbler	*Cardellina pusilla*	49
black-headed grosbeak	*Pheucticus melanocephalus*	39
black-capped chickadee	*Poecile atricapilla*	35
warbling vireo	*Vireo gilvus*	31
northern flicker	*Colaptes auratus*	30
orange-crowned warbler	*Oreothlypis celata*	28
Hammond's flycatcher	*Empidonax hammondi*	26
hermit thrush	*Catharus guttatus*	25
Nashville warbler	*Oreothlypis ruficapilla*	24
Steller's jay	*Cyanocitta stelleri*	21
dusky grouse	*Dendragapus obscurus*	12
hairy woodpecker	*Picoides villosus*	10
house wren	*Troglodytes aedon*	9
mountain bluebird	*Sialia currucoides*	9
western wood-pewee	*Contopus sordidulus*	9
sharp-shinned hawk	*Accipiter striatus*	8
American goldfinch	*Spinus tristis*	5
calliope hummingbird	*Selasphorus calliope*	5
red crossbill	*Loxia curvirostra*	5
Swainson's thrush	*Catharus ustulatus*	4
Clark's nutcracker	*Nucifraga columbiana*	3
common raven	*Corvus corax*	3
white-breasted nuthatch	*Sitta carolinensis*	3
western flycatcher	*Empidonax difficilis*	3
black-billed magpie	*Pica hudsonia*	2
Bullock's oriole	*Icterus bullockii*	2
northern pygmy-owl	*Glaucidium gnoma*	2
American kestrel	*Falco sparverius*	1
black-chinned hummingbird	*Archilochus alexandri*	1
Brewer's sparrow	*Spizella breweri*	1
band-tailed pigeon	*Patagioenas fasciata*	1
Cooper's hawk	*Accipiter cooperii*	1
northern harrier	*Circus cyaneus*	1
olive-sided flycatcher	*Contopus cooperi*	1
red-tailed hawk	*Buteo jamaicensis*	1
varied thrush	*Ixoreus naevius*	1

The best models of detection for all birds together contained factors for observer and site (electronic supplementary material, appendix S1). Observer effects were apparent for six individual species (American robin, spotted towhee, white-crowned sparrow, red-breasted nuthatch, lazuli bunting and evening grosbeak) and site effects were apparent for five species (American robin, spotted towhee, red-breasted nuthatch, Cassin's finch and evening grosbeak; electronic supplementary material, appendix S1). Detection of five species varied between noise-on and noise-off periods (dark-eyed junco, white-crowned sparrow, Cassin's finch, and chipping sparrow), and interactions between noise and site as well as noise and observer were within the best models for two species

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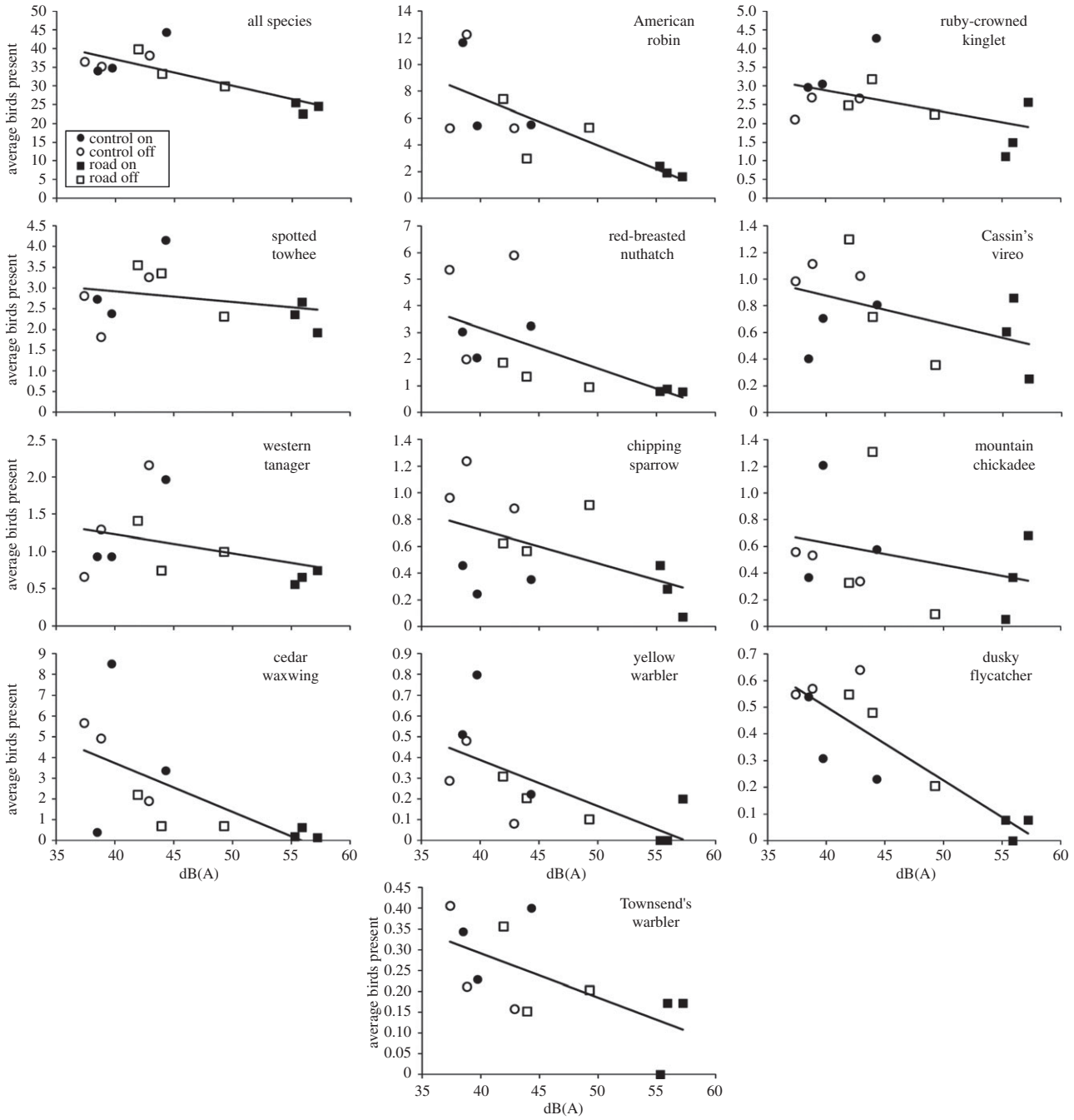


Figure 2. Average numbers of birds present per survey during noise-on and noise-off periods along the phantom road and at control sites in the Boise Foothills in southwestern Idaho. Only species with significant differences in abundance among treatments or background sound levels are shown.

(white-crowned sparrow, Cassin's finch). Finally, detection of four species varied by the day of the season (ruby-crowned kinglet, cedar waxwing, pine siskin and yellow warbler).

Abundance of every species except evening grosbeak changed as the season progressed with either linear or quadratic effects of day of the season being within models $\Delta AIC < 2$ and having confidence intervals excluding zero (electronic supplementary material, appendix S2). The best two models for the abundance of all birds within our study site received 99% of the model weight and contained a negative association with dB(A) and with the interaction between site and noise-on periods—indicating that when the noise was on, fewer birds were present near the phantom road (figure 2 and electronic supplementary material, appendix S2). Furthermore, over half of the individual species analysed responded negatively to the noise produced by the phantom road with 13 of 22 species being negatively associated either with dB(A) levels, or the interaction between site and noise-on periods, or both (figure 2). Eight species did not seem to be affected by noise because their best models were either the null or only contained covariates relating to the day of the season (electronic supplementary material, appendix S2). Finally, only one species—the Cassin's finch—was positively associated with the interaction of site and noise-on periods—indicating a positive effect of noise on abundance of this species. The number of birds present along the phantom road during noise-on periods was 28% (s.e. = 8%) lower than that during noise-off periods. Whereas, the number of birds present at the control site during noise-on periods was 3% (s.e. = 7%) higher than that during noise-off periods (figures 1 and 2). Furthermore,

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cedar waxwing and yellow warbler almost completely avoided the phantom road during noise-on periods (figure 2).

4. Discussion

This study is, to our knowledge, the first to experimentally demonstrate an effect of road noise on distributions of an animal community. To date, the most persuasive demonstrations of the effects of road noise on distributions of wildlife relied on comparing roadless areas to areas near roads [27,41–43]. Although suggestive, results from most road ecology studies are confounded by the other effects of roads such as collisions, visual disturbance and habitat alteration—and thus their interpretations have been questioned [11]. Our phantom road—an array of speakers broadcasting road noise into a roadless landscape—allowed us to isolate the effects of noise. The decline of bird abundance by over one-quarter along the phantom road, and the almost complete avoidance of two species by our treatments, suggests that road noise is a major driver of the documented effects of roads on wildlife. Therefore, our results experimentally validate the observational conclusions of past researchers—that negative effects of roads on animals can be driven by traffic noise (reviewed by [2]).

In addition, our study design of applying road noise using a line array of speakers and recording background sound levels using MP3 recorders yielded several benefits. We were able to quantify the bird community at the same sites under both noise-on and noise-off conditions instead of comparing sites that are always noisy to sites away from roads. Furthermore, our method of employing control sites, alternating between noise-on and noise-off blocks throughout the migratory period, and testing models which included covariates for day-of-season allowed us to assess and control for natural fluctuations in bird abundances and any potential differences in habitat among survey sites.

Accurate and thorough quantification of background sound levels is imperative for studies of effects of anthropogenic noise on animals [44]. Our use of MP3 recorders also allowed us to assess the LEQ over 16 days (two noise-on and noise-off blocks)—thereby performing, to our knowledge, the most thorough assessment of background sound levels in relation to distributions of terrestrial animals yet undertaken. For example, Summers *et al*. [11] quantified traffic noise over a 5 min period, Proppe *et al*. [45] quantified sound levels over four 11 min periods and several studies did not directly sample noise levels [13,41,43,44]. Because background sound level at a site can fluctuate strongly [26] and formulae estimating dB levels based on traffic volume ignore ambient sound levels, using continuous sound recordings provides a more accurate assessment of background sound levels. Our method of deploying MP3 recorders at survey sites is therefore efficient, thorough, accurate and cost effective [31].

Although our results demonstrate that traffic noise can severely affect bird abundances, other effects of roads (reviewed by [2]) will probably add to or multiply the effects of noise. Negative effects of roads are also likely to be driven by different aspects of roads depending on the taxon examined [2], and therefore taxa other than birds may be less affected by noise. Furthermore, migrating birds might strongly avoid noise because of their inherent mobility—they can easily avoid a noisy site, given there are other suitable, quieter areas nearby. Therefore, territorial breeding birds or less mobile taxa may be less willing or able to avoid noisy areas. However, our results demonstrate that noise alone is enough to cause some birds to avoid a site—suggesting that road noise might be, in some instances, the main driver of the effects of roads on animals.

Populations of migratory birds are in decline for myriad reasons including loss and degradation of migratory stopover habitat [21,23]. In fact, migration might be the most dangerous time of a migratory bird's annual cycle. For example, 85% of the yearly mortality of the black-throated blue warbler (*Dendroica caerulescens*) occurs during migration [46]. We substantially depleted the population of migratory birds and caused entire species to almost completely avoid an otherwise high-quality stopover site [25,28,47] using only traffic noise—demonstrating that anthropogenic noise can alter the amount of habitat available to migratory birds during stopover. For example, the yellow warbler—a species with declining range-wide populations [48]—was essentially absent from sites near the phantom road during noise-on periods. Because 83% of the USA is within 1 km of a road [1], it is likely that noise-sensitive species such as the yellow warbler avoid substantial areas of otherwise suitable habitat simply because they are too loud. Even within protected areas, roads can produce sound exposures similar to those produced by our phantom road across large areas [26]. Anthropogenic noise should therefore be considered when preserving and managing habitat, including stopover habitat for migratory birds.

Of course, management actions should also be informed by identifying the mechanisms underlying avoidance of noisy sites [49]. Background noise may mask important sounds, such as con- and heterospecific songs and calls, as well as sounds made by both predators and prey [10]. Increases in background noise are therefore known to increase predator vigilance in California ground squirrels (*Spermophilus beecheyi*, [50]), pronghorn (*Antilocapra americana*, [51]) and chaffinches (*Fringilla coelebs*, [52]). Because increased predator vigilance generally leads to less food intake [51,52], it is possible that noise degrades stopover habitat because migrants are less able to gain fat needed to fuel migration, but more study is needed.

An animal's ability to vocalize within frequencies which are not masked by background noise probably influences its response to increased noise [10]. For example, Francis *et al*. [7] and Rheindt [9] demonstrated that birds with higher frequency songs are less likely to avoid sites exposed to gas-compressor and road noise, respectively—probably because they are less affected by masking at lower frequencies. Furthermore, Francis *et al*. [6] demonstrated that of two closely related tyrant flycatchers, one species shifted its song to a higher frequency in response to gas-compressor noise, but did not avoid noisy areas, whereas the other species did not shift its song but was less likely to occupy noisy sites—suggesting that tolerance of noise is influenced by the ability of a species to avoid masking of its song. It is likely that the differing responses to noise among species within our study are, in part, owing to different abilities to avoid masking. However, past work has focused on masking of songs, whereas the migrating birds that we surveyed sing infrequently and to varying degrees. Future work should therefore examine the effects of masking of conspecific calls and which species possibly shift call-frequency in the presence of noise. Further, in addition to masking, other effects of noise such as disturbance,

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increased stress levels and distraction might have contributed to our results [49], and future studies should examine their relative contributions to changes in animal distributions.

Owing to logistical constraints regarding the difficulty of maintaining a phantom road, our design consisted of a single experimental and control site each containing three point count locations. That our point count locations were subsamples of the control and experimental sites might complicate the generalization of our results to other sites [53]. However, we took great care to ensure that our survey locations were as independent as possible—ridges separated survey locations and our analysis only included birds detected within 50 m. More large-scale playback studies are needed to assess the generality of our results among roads.

Generally, a deep understanding of large-scale ecological phenomena, such as those encountered in road ecology, requires both manipulative experiments and observational studies. Manipulative experiments provide strong inference into causal relationships that produce the widespread (in the case of road ecology, global) correlations demonstrated by observational studies [54,55]. Our study provides experimental demonstration of one of the primary causes underlying the correlations presented in past studies of effects of roads on animals. Future studies should employ a system similar to our phantom road to examine the effects of noise on direct measures of habitat quality such as individual fitness as well as examine the effects of noise on other taxa.

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Data accessibility. We have uploaded our data as an electronic supplementary file.

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Estimating the natterjack toad (*Epidalea calamita*) population size at RSPB Mersehead reserve using a novel identification technique

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Course: Zoology with Industrial Experience

Organisation: Royal Society for the Protection of Birds (RSPB), Scotland

Supervisor: James Silvey (Species and Habitats Officer, All Nature)

Placement dates: 4th January – 16th September 2016

Organisation

My placement was with the Royal Society for the Protection of Birds (RSPB). This conservation organisation has major influence in the UK, both on and off nature reserves, and is a partner in BirdLife International. Although its history lies in protecting birds alone, the RSPB is increasingly recognising its work on non-avian species, which is termed 'All Nature'. A Species and Habitats Officer is dedicated to All Nature in Scotland and his work is not restricted to reserve boundaries. The Reserves Ecology department, which is responsible for planning and implementing management for any habitat and species, focuses solely on reserves.

My role as an intern was split between All Nature and Reserves Ecology, undertaking projects and surveys from both. The placement was divided into three main sections: January to March was based at Scotland Headquarters (SHQ) in Edinburgh and April to June was spent at RSPB Mersehead nature reserve. The remainder was spent back at SHQ.

The natterjack toad project (All Nature) at RSPB Mersehead began in 2014, before my placement started, so there were two years of data to process in the form of photographs. No population sizes had yet been estimated, nor had any work been done with the identification software, *Interactive Individual Identification System: Spot* (I^3S). The first three months of the placement were, largely, focused on processing the data with I^3S.

My responsibilities in the second part of my placement were balanced between wardening work and the natterjack project. Approximately one third of my time was dedicated to the toad project, including monitoring spawn strings, tadpoles and water characteristics of the breeding pools. I performed the final year of surveys alongside RSPB staff and volunteers. As it was the breeding season, the wardening role was largely survey-based. This included goose counts, water rail surveys, lapwing nest and productivity monitoring, wildfowl and wader surveys, and moth trapping.

My time in the third section was split roughly in half between writing up the natterjack project and doing unrelated fieldwork. Two separate weeks were focused on the great yellow bumblebee (*Bombus distinguendus*) and its habitat in Orkney and in the north of mainland Scotland. A further week was dedicated to monitoring dark bordered beauty (*Epoine vespertaria*) on RSPB Insh Marshes. Following that, a survey was focused on searching for Irish Lady's Tresses (*Spiranthes romanzoffiana*) on the Isle of Coll, based on historical records. The final survey involved mapping string sedge (*Carex chordorrhiza*) at RSPB Insh Marshes and comparing the current distribution to that from 20 years ago.

My aim for the placement was to improve my naturalist knowledge and fieldwork skills, and develop understanding of the management process on nature reserves. The former aspect was largely achieved through the surveys, which also gave me experience in project management. Shadowing Reserves Ecologists and having the wardening role helped me to achieve the second.

Lay Abstract

The natterjack toad, which is one of two toad species in the UK, is legally protected throughout Europe. The most north-westerly population is at RSPB Mersehead nature reserve where the population size was unknown, but appeared to be diminishing. A storm in 2014 flooded the toad habitat with saltwater, which natterjack tadpoles cannot develop in.

This study aimed to estimate the size and long-term viability of the population. A series of surveys, starting in 2014, were performed to capture and recapture breeding adult males over three years. Each toad was identified by the unique wart pattern on its back. The proportion of recaptured individuals to those captured only once was used to estimate the population size. The data from each year were considered separately, giving three estimates.

Recaptures were identified using visual recognition software. Its use with natterjack toads had never been documented, so this study tested its usability and performance. Compared with recognition by eye, identifications were considerably slower and less reliable using the software. In a situation where the database is not started afresh each year, the software is predicted to save time.

The most reliable resulting population size estimate was given for 2015 with 156 breeding adult males. This number suggests that the population was not negatively affected in the long-term by the storm. In this case, it should withstand the effects of similar events in the future.

Scientific Abstract

Epidalea calamita is the second rarest of the UK's seven amphibian species. After undergoing severe range declines across Europe during the 20th century, it became a European Protected Species under the Habitats Directive (1992). A population at RSPB Mersehead nature reserve already appeared to be declining when a storm caused saltwater inundation of the habitat in January 2014. The study aimed to estimate the size and long-term stability of this population using mark-recapture models, where the permanent and unique dorsal wart patterns of *E. calamita* replaced artificial marks. The reliability and time consumption of the recognition software used (*Interactive Individual Identification System: Spot*) were also tested. Its application to *E. calamita* had not been documented before.

The surveys monitored breeding adult males and were conducted in spring 2014, 2015 and 2016. Recaptures were identified both manually and using the recognition software to test the performance of each. Each year of data was run separately through a suite of closed population mark-recapture models to produce independent population size estimates.

The most accurate population estimate was given in 2015 with 156 breeding males recorded. A population of this size should withstand stochastic events. Recapture identification was over two times slower using the software and found 79% of the matches compared to manual comparisons. The software is predicted to be faster than manual comparisons when a database is progressively added to, but its reliability with *E. calamita* should be tested further.

Introduction

Amphibians comprise over 6000 known species, with 41% in decline and 32% threatened with extinction (IUCN *et al*., 2008). Widely regarded as ecosystem indicators, they are sensitive to changes in their surroundings. Two examples among many demonstrate the use of amphibians in gauging habitat restoration success (Dixon *et al.*, 2011) and habitat disturbance (Welsh and Ollivier, 1998). However, this sensitivity makes amphibians vulnerable.

Major threats include habitat loss, primarily in the form of fragmentation (Arens *et al*., 2007; Becker *et al*., 2007), pollution (Egea-Serrano *et al.*, 2012; Polo-Cavia *et al*., 2016) and disease. The causes of decline are not mutually exclusive: one can facilitate or exacerbate another. For example, deforestation has been linked to disease emergence (Becker *et al.*, 2016). The most notable disease in recent years has been chytridiomycosis, which thrives best where temperatures fluctuations are relatively small (Murray *et al.*, 2010). Amphibians in temperate zones are, therefore, at risk.

In the UK, the intensification of agriculture during the Second World War led to loss of habitat and fragmentation (Beebee, 2014). Consequently, all seven amphibian species went into decline. A case in point is the northern pool frog (*Pelophylax lessonae*), which was presumed extinct in 1995. It was later reintroduced in 2005 to one site in Norfolk and remains the rarest amphibian in the UK.

Natterjack toads (*Epidalea calamita*) are another highly threatened amphibian species in the UK. Here, *E. calamita* underwent an 84% decline in range during the early to mid 20th century (JNCC, 2007), prompting initial protection under Schedule 5 to The Wildlife and Countryside Act (1981). Although locally widespread across Europe, declines later resulted in its designation as a European Protected Species (Annex IV to the EU Habitats Directive, 1992). Conservation efforts began in the 1970s and achieved the first successful translocation in 1980 (Buckley and Beebee, 2004). Since then, the range has expanded considerably, but the original losses have still not been recovered (JNCC, 2007).

In the UK, *E. calamita* predominantly inhabits coastal dunes, saltmarshes and heathlands (Beebee and Griffiths, 2000). Original populations were locally distributed in the east, southeast and northwest England, and in southwest Scotland (see fig. 1). In general, the northerly populations inhabited coastal systems and the southerly, heathlands.

E. calamita breeds in shallow, ephemeral pools within early successional habitats (Denton *et al.*, 1997) where vegetation is sparse and aquatic predation on larvae is low. The seasonal drying out of the pools limits the number of resident predators. These specialised requirements mean that the populations are often naturally isolated and range expansion is restricted (Buckley and Beebee, 2004).

Reasons for declines and ongoing potential threats include pollution, urbanisation and changes to water level management (McInerny and Minting, 2016). Loss of habitat due to changes in land use is thought to be the primary cause (Beebee, 1977). Hardest hit were heathland populations. This habitat requires human management to maintain it in its sub-climax state. Historical pastoral practices of

deforestation and grazing by sheep and rabbits limited vegetative growth, inadvertently cultivating ideal conditions for *E. calamita*. Scot's pine (*Pinus sylvestris*) was recognised to grow well on this land in the 19th century, so afforestation for commercial gain increased (Beebee, 1977). This and reduced grazing rights in the early 20th century contributed to the spread of pine, the loss of heathland, and, therefore, the decline of *E. calamita*.

Coastal dunes and saltmarsh have not been subject to the same pressures and do not require as much human management as heathland. Consequently, more coastal *E. calamita* populations have survived than heathland. However, they are still under threat from scrub encroachment and the continued development of coastal areas for tourism and industry.

E. calamita at RSPB Mersehead

A case in point is at Southerness on the Solway Coast, where the development led to the loss of 80% of the coastal dune, saltmarsh and agricultural land that *E. calamita* inhabited (Ellis, 1999). The toads were pushed into the reedbed at Gillfoot Bay. Though dense, this reedbed is a thin strip between the development and the sea. The development continues to encroach on this strip.

Concerns for the increasingly more vulnerable population mounted. In 1999, the Royal Society for the Protection of Birds (RSPB), Scottish Natural Heritage (SNH) and The Amphibian and Reptile Conservation Trust (ARC) translocated spawn four miles westwards to RSPB Mersehead reserve (for location, see fig. 1).

The spawn was moved into four small artificial pools arranged closely together on dune grassland (see fig. 2), inland from sand dunes. *E. calamita* becomes sexually mature at 3 years of age, so the translocation proved initially successful when spawn was found in the pools in 2002. Monitoring remained limited to spawn string counts, which can give an estimate of adult female abundance (Smith and Payne, 1980), in these four pools until 2014. The highest number of spawn strings ever reported was 11 in years 2003 and 2010, but counts dropped to 4 and 5 strings in 2012 and 2013, respectively (Silvey, 2014). Based on these counts, the adult female population was expected to consist of approximately five individuals. With such limited monitoring, however, there was an overall lack of knowledge about *E. calamita* at Mersehead.

In January 2014, a winter storm destroyed the sand dunes in which the natterjack toad hibernacula were thought to be. Without the protection of the dunes, the grasslands and surrounding merse were inundated with salt water. This increased the salinity of the toads' breeding pools. Adult *E. calamita* can withstand 25-30% seawater (3.5% saline) (Beebee, 1983). In some cases, saltwater inundation can be beneficial to the tadpoles because it kills off many of the aquatic predators. However, the tadpoles can only develop properly in less than 9% seawater (Beebee, 1983).

The effects of the storm and the former decline in spawn string numbers prompted this three year study, the aim of which was primarily to estimate the size of the population. The secondary aim was to assess whether the storm of 2014 had been detrimental to it. An isolated population, such as that at

RSPB Mersehead, should consist of more than 250 adults (both males and females included with ratio of 1:1) to withstand stochastic events (Buckley and Beebee, 2016). At this size, the normal fluctuating nature of *E. calamita* productivity (Baker *et al.*, 2011) can mitigate the effects of these events. The study took this figure as a baseline: if the population size estimates were below this, the storm and any similar future events had been, and would be, detrimental, and vice versa.

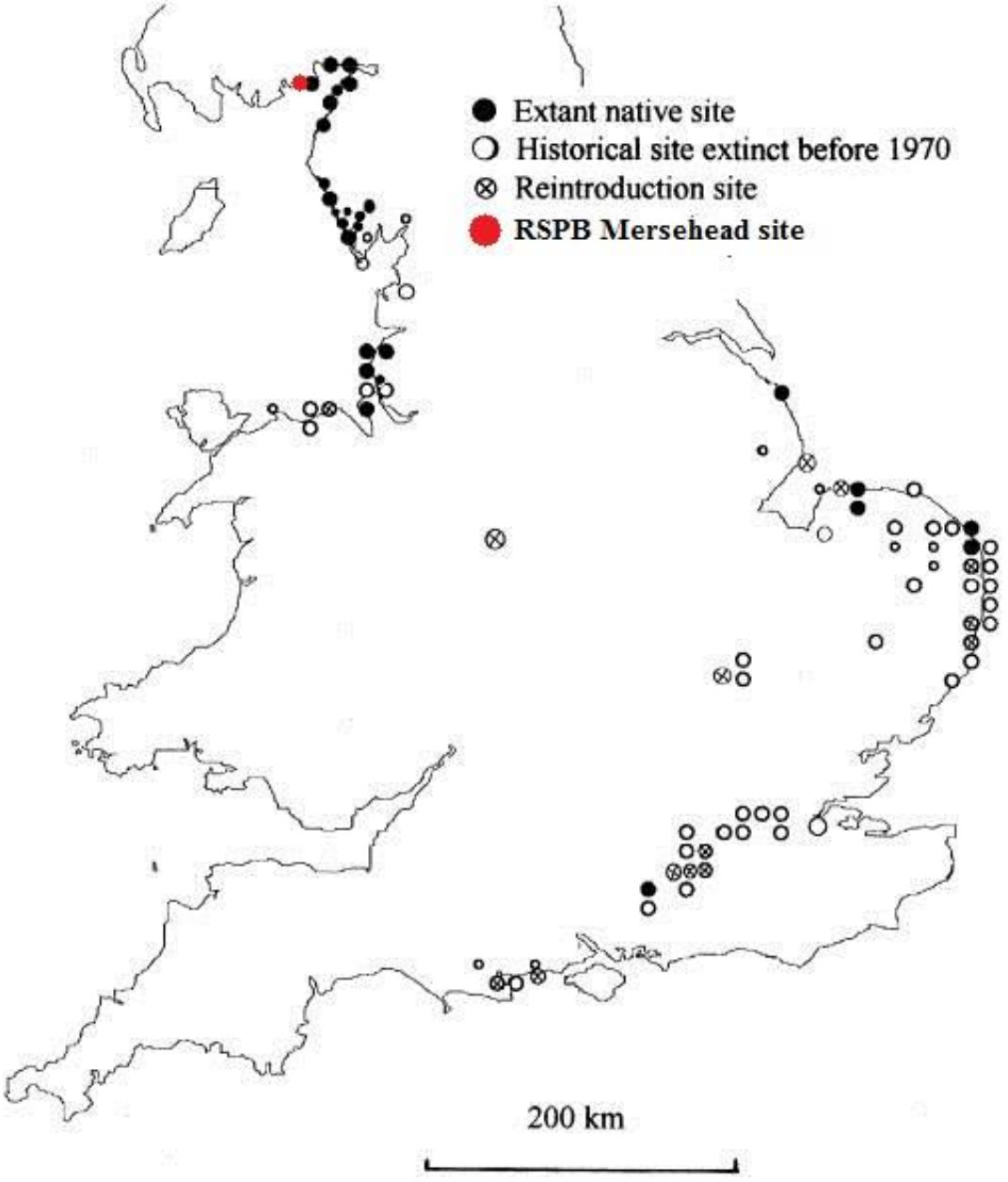


Figure 1. Map of UK *Epidalea calamita* population locations. The map shows original populations, both extant and extinct, and sites where *E. calamita* has been reintroduced or translocated to. The RSPB Mersehead reserve at the most north-westerly point is highlighted.

Figure adapted from Denton *et al.* (1997).

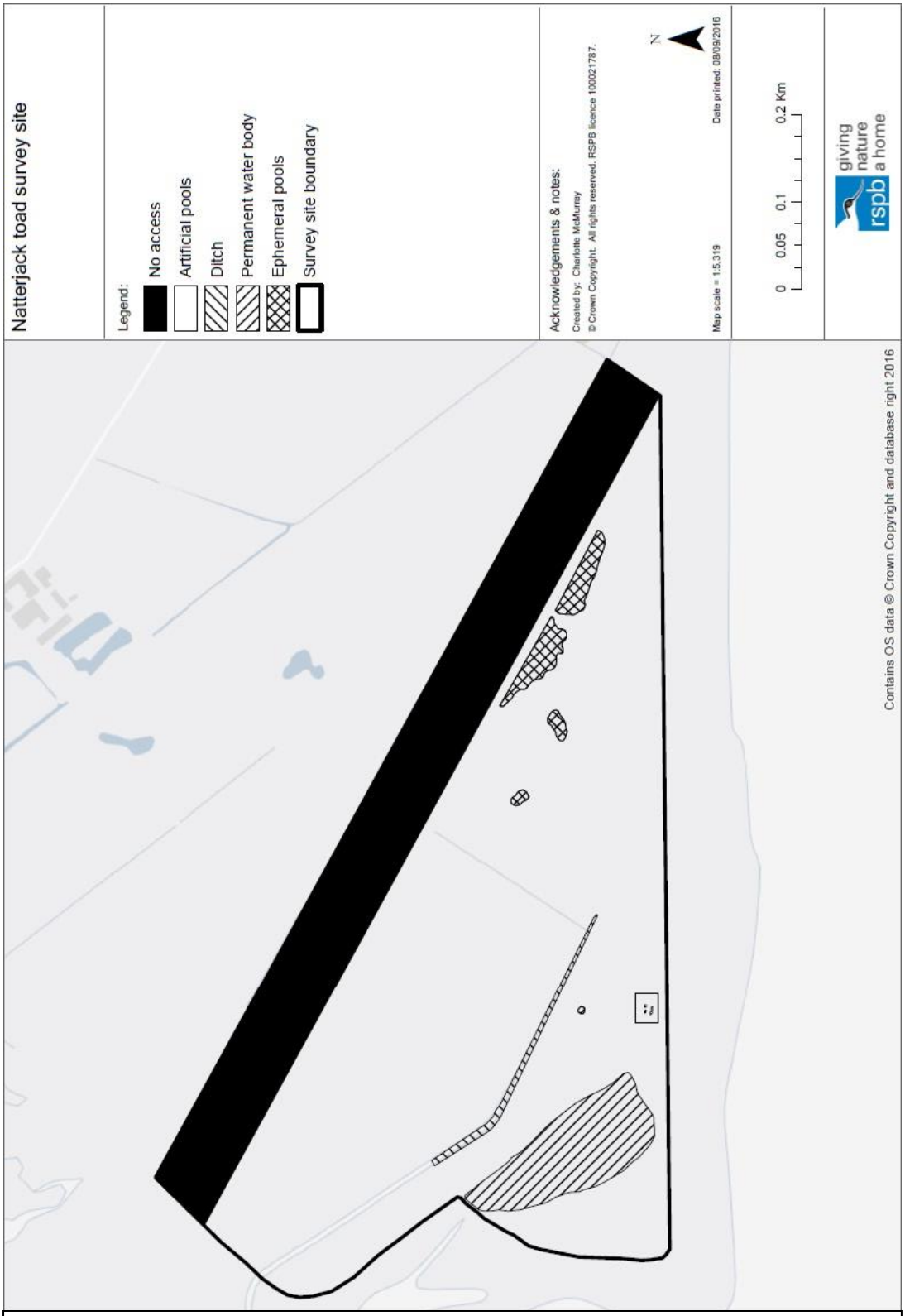


Figure 2. Diagram of breeding areas within survey site. The breeding areas that can be accessed comprise five natural pools, a larger, permanent body of water to the south west of the site and a ditch. Four small artificial pools are on the south side. In the area with no access, there are deep ditches and scrub.

Survey method options and novel identification technique

Traditional methods for monitoring *E. calamita* include counting toadlets as they emerge from the breeding pools en masse or noting the number of spawn strings produced in a season. The former is suggestive of how successful the breeding season was and of the recruitment for that year. Neither method was an option on this survey site beyond the four artificial pools because the vegetation, both surrounding and within the pools, is too dense in some areas. Also, some of the pools are expansive, so spawn string surveys would be inefficient. If attempted, it is likely that more strings would be damaged than counted.

Furthermore, spawn string counts are not resilient: they assume that all strings are whole, which is not always the case, and that a female will produce only one spawn string within a breeding season. In actuality, some spawn twice (Beebee, 1983) or not at all (Buckley and Beebee, 2004).

As traditional methods were not appropriate in this system, mark-recapture (MR) surveys were trialled to estimate the population size. Another aim of this study was to demonstrate that MR can be an informative alternative, giving data at an individual specimen level. Traditionally, individuals are captured and marked in a unique way during the first survey. This can range from fitting leg rings to birds, to paint-marking insects and toe-clipping amphibians. In any subsequent survey, recaptures of the originally-marked individuals will likely be made, in addition to unmarked specimens. The ratio of new captures to recaptures is used to estimate the size of the population. In general, the smaller this ratio, the larger the population is. Bird-ringing is a good example of large-scale, long-term MR surveying. It is best known for the data on migration that it can produce (Marra *et al.*, 2005; Visser *et al.*, 2009).

Although toe-clipping in amphibians is widely used for MR surveys, there is much debate about its ethics and any impacts on the data collected. It has been argued that it can affect the recapture rates of specimens (Perry *et al*., 2011; McCarthy and Parris, 2004; Davis and Ovaska, 2001), either by increasing their chance of death by infection, of predation by increased visibility, or by changing their behaviour. The results of any survey would be, therefore, skewed in favour of a seemingly larger population than in actuality. Other studies have shown it to have minimal impact, if any, except where it affects burrowing behaviour (Phillot *et al.*, 2007). Since *E. calamita* commonly excavates its own burrows, toe-clipping was not an option for this study.

Instead, biometrics were relied on. Biometrics are increasingly being used for individual animal identification. Most animals have unique variations in their appearance that do not change over time, so these characteristics can be used to recognise individuals using photographs alone. For example, individual elephants (both *Loxodonta* and *Elephas* genuses) can be identified by the shape of the ears (Ardovini *et al.*, 2008). It has also extended to animal colouration patterning, as with cheetah (*Acinonyx jubatus*) (Miththapala *et al.*, 1989), humpback whale (*Megaptera novaeangliae*) (Katona *et al.*, 1979) and African penguin (*Spheniscus demersus*) (Burghardt and Campbell, 2007).

E. calamita is another suitable candidate because the dorsal wart patterns are unique to each individual and do not change throughout their lifetimes (Arak, 1983; Minting, 2012a).This non-invasive capture-recapture (CR) method is suitable for estimating population size using basic MR models.

Using biometrics to identify recaptured individuals can be highly time-consuming, especially with large populations. Development of individual recognition software is ongoing and it is facilitating the use of the use of biometrics by shortening the length of time it takes to process photographs. Studies have also expressed concern for the reliability of identifying individuals by eye alone (Speed *et al.*, 2007) or by alternative methods, such as visible implant elastomers (Bendik *et al.*, 2013). These studies have found software to improve the match reliability.

As far as is documented, recognition software has not been used with *E. calamita* before. The software selected was *Interactive Individual Identification System: Spot* (I^3S). It had already been successfully used with other species, including spotted ragged-tooth shark (*Carcharias taurus*) (van Tienhoven *et al.*, 2007), whale shark (*Rhincodon typus*) (Speed *et al*., 2007), the irises of *Tarentola* geckos (Rocha *et al*., 2013), Italian crested newts (*Triturus carnifex*) (Sannolo *et al*., 2016) and *Rosalia alpina*, a cerambycid beetle (Caci *et al*., 2013). These studies found that match success could be roughly 95% with markedly different species, from *C. taurus* (van Tienhoven *et al.*, 2007) to *R. alpine* (Caci *et al*., 2013). However, the angle at which the photographs are taken in relation to the subject affect this success rate (Speed *et al.*, 2007).

These CR surveys gave an opportunity to test the reliability and time consumption of I^3S with *E. calamita*. It was hypothesised that the software would produce promising results, as in the identification studies above.

Hypotheses

1. **The population would consist of approximately 10 adults (both males and females).** This is based on the spawn string counts of the preceding years and a 1:1 male to female ratio.
2. **The storm of 2014 was detrimental to the population**. This will be assessed by comparing the population size estimates to the hypothetical stable population size of more than 250 adults (Buckley and Beebee, 2016).
3. **The CR method will successfully replace of other survey methods for *E. calamita*.** The method will provide a reliable monitoring option where the habitat limits others, like counting spawn strings or toadlets.
4. **I^3S will save labour time and improve match reliability over manual comparisons**. This will either support or negate the use of *E. calamita* biometrics in future CR surveys.

Materials & Methods

Study site

The study site is a 35 hectare dune grassland section of RSPB Mersehead reserve, NX 923 553. This is, as far as is known, the breeding range of *E. calamita* on this reserve. Within the site are five shallow ephemeral pools, a larger, permanent water body, a ditch and four small artificial pools (see fig. 2). Part of the site consists of deeper ditches and scrub, but this could not be accessed to survey. It is thought that *E. calamita* breeds there, as multiple males were heard calling in 2015 and 2016.

Survey method

The CR surveys, performed in conjunction with ARC, ran each year from 2014 to 2016. Each year was analysed separately. Although not a participant in the first two years of surveys, the author was involved in all other aspects of this study. Surveys were carried out three times each year within the *E. calamita* breeding season. The three survey occasions were expected to give enough recapture data for reliable population size estimates to be made for each year, whilst keeping disturbance to a minimum. The surveys were spaced a week apart.

Males and females behave differently in many species. In amphibians, it is common to eclude females in surveys because male behaviour is more consistent throughout a season. This is the case for *E. calamita*. The males return to the breeding pools every night to call to females, whether they have already mated or not. Females, however, only frequent a pool on the night they are ready to spawn. This may only happen up to twice a season. Therefore, the population estimates produced by the MR models in this study are for adult breeding males only.

The breeding season of *E. calamita* at the study site varies, but starts between late March and early April, and ends in early to mid-June. The first survey of each year was performed when the breeding seasons started, which was signalled by the constant or ‘full’ chorus of the male toads at night. Calling is intermittent prior to this, so nightly checks were made from mid-March to confirm whether the season had truly begun. In both 2014 and 2015, the first survey was held on 23rd March, while in 2016, the first was on 3rd April.

The weather conditions affect the number of male toads that call. The most suitable date each week was selected according to the most favourable weather conditions. This is when the night-time air temperature is over 9°c and there is substantial cloud cover (McInerny and Minting, 2016).

Males start calling approximately 1.5 hours after sunset. This marked the start of each survey. Therefore, the start time became progressively later over the three surveys. For example, in 2016, the surveys began at 2130, 2200 and 2230 respectively. The chorus peaked then tapered off two hours after it started. This was a natural end to each survey.

Sampling effort was maintained as much as possible within years. However, there were some minor changes in survey effort between years due to changes in logistics: one group of three to five people

sampled in 2014 and 2016, but two groups recorded in 2015, in which year toads were more abundant. All recorders were appropriately licensed by SNH to handle and photograph *E. calamita*.

At the start of each survey, the calls were used to locate males within the compartment. With the use of flashlights, toads were easy to spot in the breeding pools. The vast majority were captured by hand, but some individuals were quick to bury themselves in the pool sediment and could not be retrieved. All captures were sexed: male throats have a blue and red tinge (see fig. 3) and the toes on their forefeet have visible nuptial pads, while females have neither. Each individual was assigned an identification number and their dorsal wart pattern was photographed with a Canon Powershot G16 camera in 2016. The following settings were most appropriate: ISO 800, aperture 5.6 and shutter speed 1/160. The camera type and settings were not recorded in the previous two years.



Figure 3. Photographs of male and female throats. a) Breeding male natterjacks have characteristic blue and red throats. b) female natterjacks do not have any blue or red throat colouration.

Figure 3a from Roseanne Watt; figure 3b adapted from (McInerny and Minting, 2016).

In 2014 and 2015, the photographs were taken against a laminated white sheet, which allowed the subjects to adopt any posture (see fig. 4a). There is a higher proportion of subjects running rather than sitting in the 2015 database than in that of 2014: 33% to 8%. This correlates with a higher identification success rate when utilising the identification software (see 'Identification technique development' section in 'Results'). Therefore, the seated posture was taken as the standard posture.

To encourage this position and limit movement in 2016, the sheet was exchanged for sponges with cuboid scrapes cut out of them for the toads to sit inside (see fig. 4b). The dimensions of a complete sponge were 11 x 7 x 3.5 (length x width x depth, cm); the inside scrape was 8 x 4 x 2.5. The sponges were pink to give substantial contrast to the toads. The sponge technique did not work for amplectant pairs, so the photograph was taken when the pair was simply held by the handler. The animals were then returned to their respective breeding pools.

The process took approximately five minutes for each toad, so disturbance was minimised. If amplectant pairs were spawning, they were not captured, but their presence was noted. For the development of the photography technique and identification method, data were gathered for females and for any individuals who could not be sexed reliably. These data were discounted from the statistical analyses.

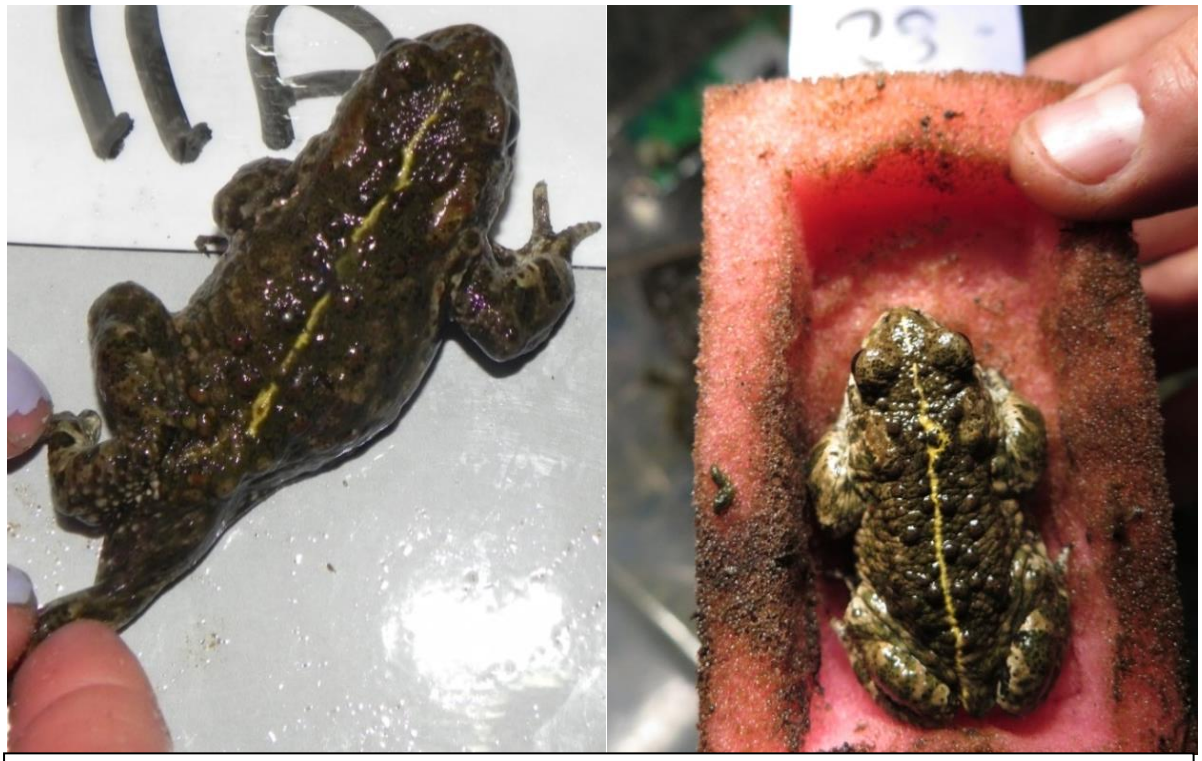


Figure 4. Demonstrations of varying postures and materials used in different years. a) Photographs were taken against a laminated white sheet in 2014 and 2015. Movements of the subject were not restrained, so various postures could be adopted, like the 'runner' shown in the figure. b) Sponges were used to restrict the movement of toads in 2016 to encourage the standardised sitting posture shown in the figure.

Identifying recaptures

Photographs were compared using the identification software, I^3S, to identify recaptures. I^3S requires manual input of the data. To create a database from the photographs, the user selects three reference points on the specimen; the 'elements' or characteristic permanent markings can then be highlighted. In this study, the reference points on the toads were the armpits and the vent, and the elements were the toads' warts. The developers of I^3S, den Hartog and Reijns (2014), recommend selecting a maximum of 30 elements and a minimum of 10 to give the best results. If a specimen has more than 30 warts, only the most prominent were selected. This creates a 2-dimensional pattern for each photo (see fig. 5). To ensure the reference points in each pattern 'match up', the software compares them using 2-dimensional affine transformation (van Tienhoven *et al*., 2007).

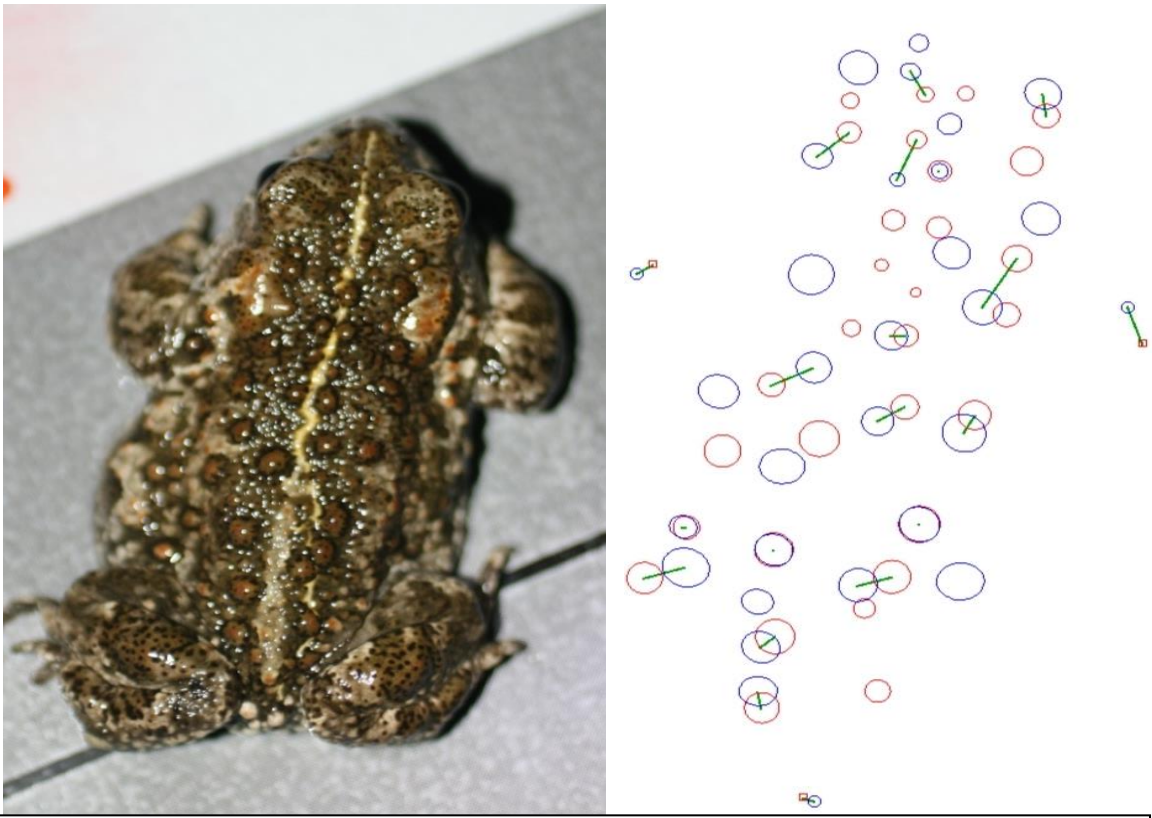


Figure 5. Creation of the 2-dimensional pattern by I^{3}S from the toad photograph. Three reference points are represented by red squares. The warts of the individual in the photograph are represented by red circles. The blue circles show the wart pattern of a different individual. Green lines and dots show which warts are matches in the patterns.

Figure on right adapted from I^{3}S software.

The software considers different ways to match the elements of one pattern with others. For each element that matches another in a different fingerprint (a matched pair), both the size difference of the elements and the distance between them are used to produce a match score. The lower this score is, the closer the match. Bigger differences in element size generate higher scores. The distance is incorporated into the score by dividing the sum of the distances between each matched pair in two fingerprints by the square of the element number (van Tienhoven *et al.*, 2007).

Once inputted, a photo can be compared to multiple photos either from the rest of the same database or from another. The software then produces a ranked list of fifty possible matches (if the database is big enough), from which the user can make the final decision.

This study considers a match to be 'found' when it is in the highest 10 rankings. This is based on the finding that 92% of matches are recognised within these rankings (den Hartog and Reijns, 2014). In this case, however, these databases consisted of only one photo per individual instead of two. Matches listed between positions 11 and 50 have still been noted in this study.

The photograph databases for the three surveys within each year were compared. The whole process, from inputting the photographs to searching the ranked list, was timed for the 2016 database only. The timings for the previous two years' data were too approximate, so could not be used.

Photographs were also compared manually (by eye, without software) to test the reliability and time consumption against I^3S. Three people assessed the 2014 database to test consistency; the time demands to do the same for 2015 and 2016 were too great, so only one researcher did so.

Statistical analysis

The survey method was designed to be applicable with closed population MR models. The assumptions of these models are such (White *et al*., 1982):

1. There is no emigration, immigration, birth or death
2. All individuals have equal catchability
3. Marking does not affect survivability or recapture
4. Markings are not lost or misrecorded

These assumptions have been met as closely as possible. Every year has been analysed separately and the three survey nights within each were distanced by one week. This limited the chance of stochastic events, predation and movement. It is thought that there is no migration between any neighbouring populations. This belief is based on the lack of suitable habitat surrounding the survey site and the negative results of auditory surveys in neighbouring areas. Births were not applicable because *E. calamita* becomes sexually mature at three years old, after which time it breeds for the whole season.

As the survey considered adult males only, the behaviour of all individuals should have been similar. Therefore, they should have had similar catchability. Mortality or changes in behaviour caused by marking animals was avoided because there was no need to do so. Moreover, the marks could not be lost and the photographic method minimises the risk of misrecording any individual.

The survey method allowed identification of toads at the individual level. This made it possible to construct a capture history for each male, where a capture on a given survey occasion is marked as '1' and no capture as '0'. With three surveys in a single year, this could become '100', for example (see table 1).

In the fictional example above and following the assumption that death will not occur in a closed population, this male was captured once then avoided recapture twice. This suggests that the recapture probability is low in comparison with a hypothetical capture history of 111. Therefore, there are two probabilities to consider in a closed population: the probability of capture (p) and that of recapture (c). These probabilities for each year are estimated by the models from the capture history data. Using this information, density (f_0) is also estimated, from which the size of the population (N) can then be derived.

For each year individually, a suite of traditional MR models, developed by Otis *et al*. (1978), were run (see table 2 where p = capture probability, C = recapture probability and f_0 = density). This suite included models which held the probabilities of capture and recapture constant throughout the study period, and others which allowed these probabilities to vary between the three capture occasions. By considering various models, the most appropriate descriptor of each year's dataset could be selected.

The models were run using Program MARK (White *et al*., 1982) with R statistical language (R Development Core Team, 2008), through R MARK (Laake, 2013). The most supported model for each year was selected according to its Akaike Information Criterion (AIC) value (Akaike, 1974). This method allows selection of the 'best' model – that which describes the data in as few parameters as possible. The model with the lowest AIC value is the most parsimonious explanation.

The decision was also informed by the relative percentage support for each model, where support of 50% or above is significant. The model with the lowest AIC value and the most support from the suite was considered to be best fit for the data. This model was used to produce the population size estimate for each year. The results were then visually assessed using Microsoft Excel.

Identification number	Capture event 1	Capture event 2	Capture event 3	Capture history
1	1	0	1	101
2	1	1	1	111
3	0	1	1	011
4	1	1	1	111
5	1	1	0	110
6	1	1	0	110
7	1	1	0	110
8	0	0	1	001

Table 1. A sub-section of individual capture histories from 2014. Each toad is identified by a number, shown in the left-hand column. Captures are represented by '1' and no capture by '0'. The right-hand column summarises the three capture events.

Model	Formula	Number of parameters	Description
Constant Model	p(~1) = c(~1); f_0(~1)	2	Probabilities of captures and recaptures are equal and constant over time.
Probability-varying Model	p(~1), c(~1); f_0(~1)	3	Probabilities of captures and recaptures are not equal ($p \neq C$), but each remains constant over time, e.g. $p_1 = p_2 = p_3$.
Time-varying Model	p(time) = c(time); f_0(~1)	4	Probabilities of captures and recaptures are equal ($p_2 = C_2$ and $p_3 = C_3$), but these vary over time ($p_1 \neq p_2 \neq p_3$).
Varying Model	p(time); c(time); f_0(~1)	6	Probabilities of captures and recaptures are not equal and all vary over time.

Table 2. The suite of basic closed population models used. The table describes the suite of closed population models chosen to best explain the data, where p = capture probability, C = recapture probability, f_0 = density. Parameters are separated by ';' and ~1 means 'is constant'.

Results

Captures and recaptures

There were 47 capture events in total from the three surveys in 2014. Eight of these were either female or of unknown gender and photographs were missing for a further 8 male toads (see fig. 6 for usable photograph proportions). These 16 captures were discounted. Of the 22 adult males with photographs, 5 were recaptured once and 2 recaptured twice (see fig. 7).

The number of toads present during the surveys was much higher in 2015, when 105 adult males were captured, 25 of which were recaptured once and 6 recaptured twice (see fig. 7). There were 178 captures in total, but 36 were discounted: 22 were either female or of unknown gender and 14 photographs of males were of too poor quality (see fig. 6).

In 2016, there were 122 captures in total, 13 of which were either female or of unknown gender, and 1 was a juvenile male. Photos were not taken of three amplectant males this year to limit their disturbance. Of the remaining 92 adult males captured, 14 were recaptured once and none were recaptured twice (see fig. 7).

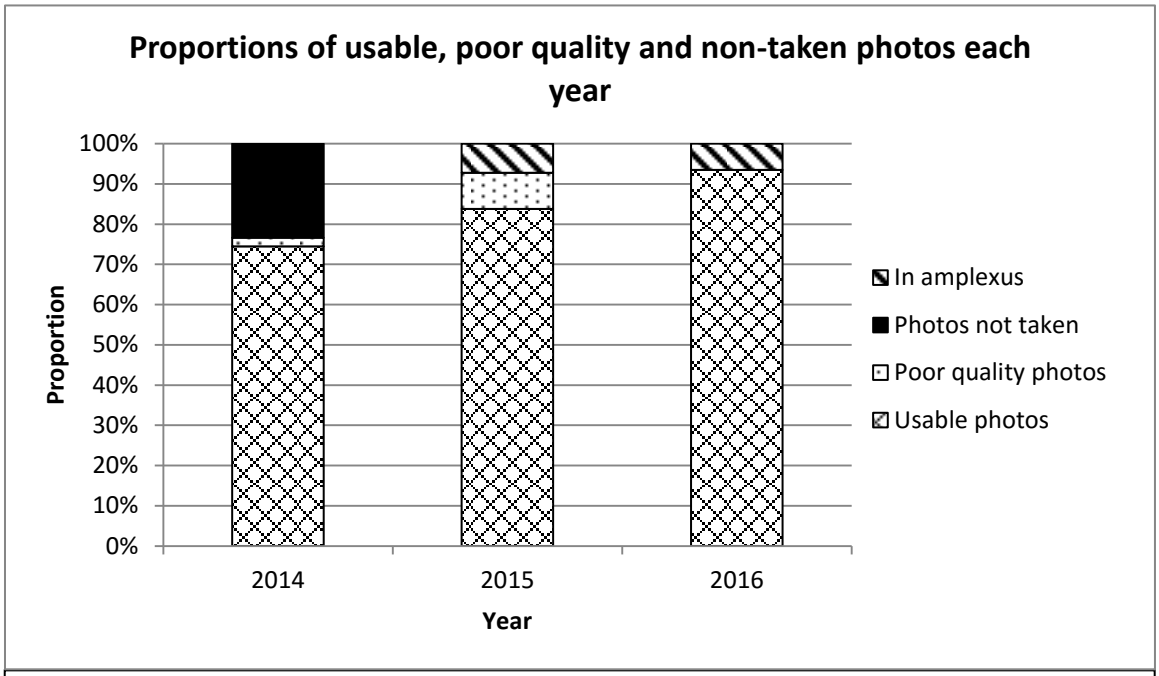


Figure 6. Proportions of usable, poor quality and photographs not taken each year. The proportion of usable photographs improved each year. Between 2014 and 2015, the number of photographs not taken diminished to 0%, but the proportion of poor quality photographs increased. The proportion of photographs not taken was maintained into 2016 and all individuals were recorded by photograph. Amplectant individuals could not be photographed.

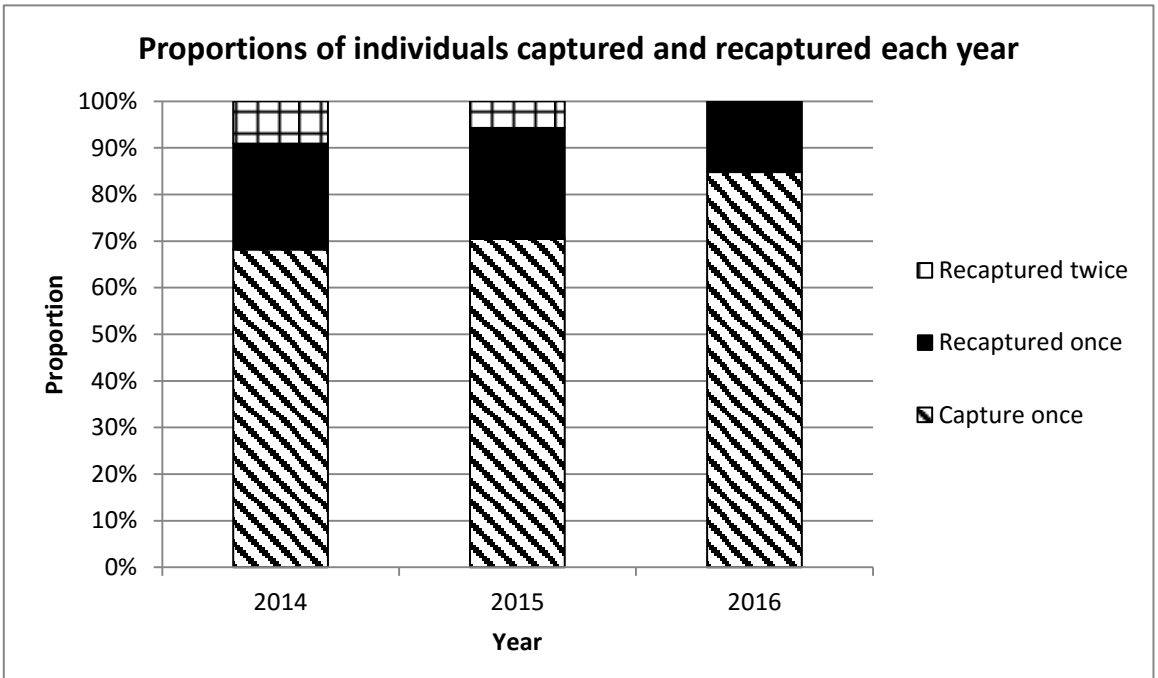


Figure 7. Proportions of individuals captured and recaptured each year. The proportions of toads captured once was maintained between the first two years and those recaptured only once increased by 2%. In 2016, over 14% more toads were captured only once than in 2015 and none were recaptured twice.

Statistical analysis

The most suitable models according to the lowest AIC values were the Constant Model for 2014 and the Time-varying Model for both 2015 and 2016 (see table 3). These selections each had over 50% support.

The breeding male population in 2014 was estimated to consist of 30 individuals, with a confidence interval of 24 to 51. The 2015 data supports a population of 156, between 133 and 196. For 2016, 248 males were predicted, with lower and upper limits of 173 and 394 respectively (see fig. 8).

Identification using I^3S

Seven out of 9 matches (78%) were found from the 2014 photographs (see table 4). The remaining two were ranked 17th and 29th, but the database was small, consisting of 32 individuals. In the 2015 database (146 photographs), 49% of matches found – 18 out of a possible 37. Six of those not found were ranked between 17th and 30th, but the remaining 13 were not listed at all.

No matches in either year were found by I^3S that had not been found manually. All three manual assessors found exactly the same matches within 2014 as each other.

	Model	Number of parameters	ΔAIC	% support
2014	Constant	2	0	58%
	Probability-varying	3	1.6	26%
	Time-varying	4	3.2	12%
	All-varying	6	5.3	4%
2015	Time-varying	4	0	66%
	All-varying	6	1.3	34%
	Constant	2	15.0	0.04%
	Probability-varying	3	15.9	0.02%
2016	Time-varying	4	0	66%
	All-varying	6	1.9	25%
	Constant	2	4.8	6%
	Probability-varying	3	6.6	2%

Table 3. Details on model selection for each year of data. The thick borders denote the most supported model and associated parameters chosen for each year. The order of models listed correlates with the relative suitability of each. ΔAIC demonstrates the differences between the AIC values of the selected model with the others. The support for all selected models is over 50%.

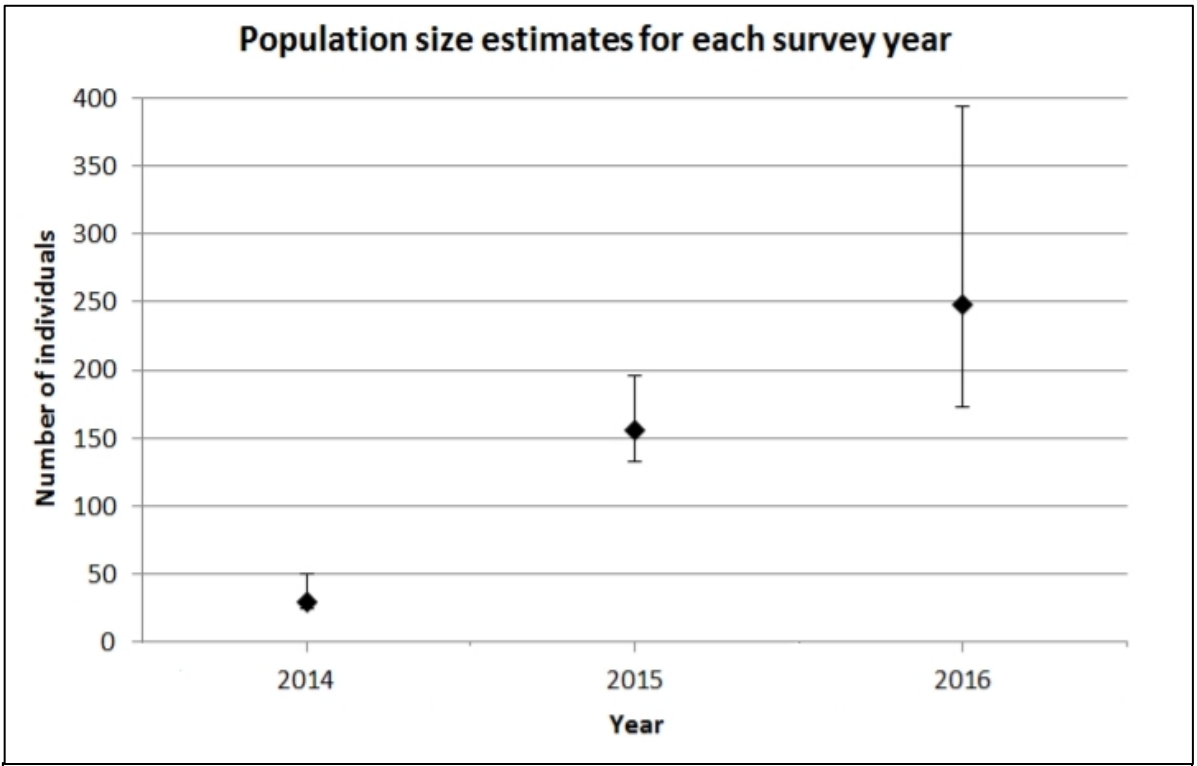


Figure 8. Population size estimates for each survey year. The estimates given for 2014, 2015 and 2016, respectively, are 30, 156 and 248. The confidence intervals for 2014 are 24 to 50; for 2015, it is 133 to 195; and for 2016, the lower limit is 173, while the upper is 394.

	Proportion of matches found, ranked and not ranked		
	1st-10th (match success)	**11th-50th**	**Not ranked**
2014*	78% (1st-2nd)	22% (17th-29th)	0%
2015*	49% (1st-10th)	16% (17th-30th)	35%
2016	79% (1st)	21% (17th-19th)	0%

Table 4. Matches found, ranked and not ranked by I^{3}S each year. The proportion of matches found (those ranked between 1st and 10th) is significantly lower, in 2015 than in the other two years, while those not ranked at all was the highest. Both 2014 and 2016 have similar proportions, but all of those found in 2016 were ranked 1st, while they were 1st and 2nd in 2014. The remainder of the photos for 2016 were ranked higher (17th to 19th), as opposed to 17th to 29th in 2014.

Identification technique development

The best results were produced by I^3S when the toad posture was standardised by the sponge technique and the dorsal wart pattern was parallel to the camera lens. Photograph quality also impacted on whether the wart patterns could be clearly deciphered, so the night-time photography technique was improved in 2016.

To support these findings, the database for each survey was narrowed down to include only the most suitable photos according to the above criteria. This reduced the sample sizes considerably, but improved the match success rates. The updated 2014 database could include only 45% of the original, but produced a 100% match success, using the manual comparison results as the baseline. Just 37% of the 2015 photographs could be used and, with one match unidentified, these produced an 86% match success.

In 2016, every photo could be used because of the standardised toad postures and the improvement in photograph quality (see fig. 6). I^3S found 79% of the matches (11 out of 14) that were identified manually, without any new finds. Unlike the database searches from the other years, all matches, if found, were ranked first on the list. The matches from 2014 and 2015 ranked from 1st to 2nd and 1st to 10th respectively. The remaining 3 matches for this database of 109 were 17th, 18th and 19th, in better positions than their equivalents for the other years.

With regard to time consumption, the manual comparisons were the quickest, taking three hours (3h00) to complete. I^3S took 6h30, 4h05 of which were used to input the data. This was a rate of 2.3 minutes per photo for input and 1.7 minutes per photo for comparisons.

Discussion

Hypothesis 1: the population would consist of approximately ten adults (both males and females)

The population size estimates for each year, respectively, were 30, 156 and 248 adult breeding males (for confidence intervals, see fig. 8). These figures are the first ever derived from the population at Mersehead. Each one of these estimates disproves the first hypothesis, which was based on spawn string counts and a 1:1 male to female ratio.

The figure for 2016 was far higher than expected. This may have been caused by the lower recapture rate during these surveys. In comparison to 2015, there were 10% fewer recaptures and no individuals were caught on all three occasions in 2016. As many previously unseen males were being captured, the models struggled to predict the upper limits of the population size. In reality, the recapture probability simply was not consistent. The weather conditions for the second survey in 2016 were unfavourable (high wind speeds), so there were fewer males calling than during the other two surveys that year. As a consequence, the likelihood of recapturing individuals was lower this year than in 2015.

Moreover, the survey effort in 2015 was double that of 2016, with two groups of people recording rather than one. In the future, effects of bad weather can be mitigated by increasing the monitoring effort, in terms of number of surveys in a given year. The number of groups during each survey should be consistent both within and between years.

The breeding male estimate is more than five times lower for 2014, the year of the storm, than for 2015. *E. calamita* productivity is renowned to be 'boom or bust' from year to year (McInerny and Minting, 2016). Therefore, this difference could have been due to a less successful breeding season in 2011 than 2012. The effects would then be apparent in these surveys three years later when the offspring had matured. There are no appropriate records from 2011 and 2012 for comparison because only the four small artificial pools were monitored. At this time, the natterjack toad range at Mersehead was unknown.

There are two other possible explanations for this. The first is that a large number of toads died in the storm, but the numbers were replenished by immigration from neighbouring populations. This is unlikely because the habitat surrounding Mersehead is unfavourable and auditory surveys performed in these areas gave negative results.

The second is that the toads simply did not breed in 2014 because of saline conditions in the breeding pools (ranging from 0.6% to 3.2% salinity). Courtship and breeding have high energy costs and, in saline water, there is great risk that the tadpoles will not survive (Beebee, 1983). These risks are likely to have outweighed the chance that the males would not only mate, but that the young would survive. It was noted that no amplectant pairs were found in 2014, while 13 and five were found in the subsequent respective years. However, this support is far from robust.

Hypothesis 2: the storm of 2014 was detrimental to the population

This hypothesis was tested by comparing this study's estimates to the ideal stable population size of more than 250 adult males and females (Buckley and Beebee, 2016). Using the 1:1 male to female ratio and presuming that the females responded to the storm in the same way as the males, there were 60 breeding adults in 2014. This rises to 312 for 2015. The 2016 estimate is not considered because of the concerns for its reliability. Despite that, it is known that the smallest the breeding male population could be is 92 individuals because of the number captured during the 2016 surveys.

The new findings satisfy the population size ideal of more than 250 adults in a 'normal' year without any stochastic event (Buckley and Beebee, 2016). Therefore, the storm should not have been detrimental to the long-term health of RSPB Mersehead's population. However, these estimates are given with caution because of the lack of consensus on the male to female ratio. Beebee (1983) even proposes that this ratio varies with latitude. Focused studies should be performed on this to allow better assessment and understanding of *E. calamita's* populations. Additionally, the frequency of bad breeding years at Mersehead must be recorded. Causes should be noted, from major events to more subtle ones, like pools drying out before the toadlets can emerge.

Hypothesis 3: the CR method will successfully replace other survey methods for *E. calamita*

Reliably estimating the size of populations is crucial in assessing how any recovery of *E. calamita* is proceeding. The spawn string counts before 2014 at RSPB Mersehead highlight how limited the method, on which much of the UK data is based on (Beebee and Buckley, 2014), can be. The spawn string counts at Mersehead before 2014 were performed in daylight when toads were not calling, so there was no obvious indication that their range had expanded. The records instead intimated that the population was in decline. This survey has found that preliminary auditory surveys provide an easily-obtained and useful baseline, identifying how widespread the male toads are.

The major benefit of the CR method is that data can be gathered at an individual level. It is crucial in monitoring population aspects, such as size or dispersal. The use of the dorsal wart patterns as identification bypasses the potential risks of marking specimens. These risks can include uncertainties, like the loss or change of marks over time, or whether the behaviour or survivability of the individual is affected. Thus, the assumptions of a closed population are met and can be modelled confidently.

A drawback of the technique is the interruption of breeding activities, although this is minimal per toad. Also, it is unknown how the females respond to such disturbance and whether it affects their mate choice. Until this aspect is known, surveys of this intensity should not be performed frequently with smaller, more vulnerable populations.

Despite the drawback, the CR method was a successful replacement of other survey methods in this study. It is recommended for other studies, particularly in similar habitats to those at RSPB Mersehead.

Hypothesis 4: I^3S will save labour time and improve match reliability over manual comparisons

The identification software, I^3S, did not satisfy the last hypothesis either in reliability or in time consumption for any year, even after the improvement in technique in 2016. Manual comparisons were still faster and more reliable. The match results did, however, improve markedly in the final year compared to 2015 and were marginally better than 2014 (see table 4). Both reliability and time considerations are discussed further below.

Reliability

The match success rate for the 2016 data (79%) was not as high as expected. This may be due to subjectivity when selecting the most prominent warts for the creation of the 2-D pattern. One of the missed matches from 2016 was due to different warts being selected from the two photographs. The standardisation of photographs does counteract this to an extent, but further refinement of the technique is unrealistic in the field.

One downfall of this study is that only one person tested the software, so the degree of user error is unknown. I^3S should be subject to further testing using the same photography techniques as in 2016.

Firstly, any effects of user subjectivity need to be assessed. Secondly, more information on whether the match success rate remains static, correlates with the size of the database or if it is unpredictable must be gathered. A firm decision about the use of I^3S in *E. calamita* CR studies can then be made.

Time consumption

In terms of labour time, it took 6h30 to use I^3S, yet 3h00 to perform manual comparisons. Despite this, if the software was used to compare across years, most of the database would already exist from previous years. Data input would not be as timely as if the database was started from afresh each year. As the database grows, the time difference would be reduced, as Rocha *et al.* (2013) also found, and manual comparisons would become the slower of the options.

For example, if the photographs from all three years in this study had been suitably comparable, it would have taken 9h05 to use I^3S. This is based on the rates of use from within-2016. The surveys with fewer captures were also compared to those with more for efficiency. For instance, the survey with 32 was compared to those with 146 and 109, then 109 to 146. The comparisons across years that were performed manually actually took 15h45 – a time difference of 6h40. This margin would only increase over time as the database grew.

To conclude, I^3S is not appropriate for use when the number of photographs to be inputted equals the number to be compared, but it is the favoured option as the number to be inputted becomes proportionally lower.

Conclusions

The Mersehead population size is far larger than anticipated, based on spawn string counts pre-2014, and shows promise of being resistant to future stochastic events according to Buckley and Beebee (2016). Assuming that the female estimates, derived from the male figures, are correct, the storm of 2014 should not have been detrimental.

Spawn string counts are inadequate in habitats with lots of vegetation and large pools, as at RSPB Mersehead. Listening at night is essential to establishing the range of the male toads. CR methods using MR models provide a reliable way of assessing population and behavioural information about individuals. The use of the dorsal wart patterns as identification is preferable to marking because some assumptions of closed population MR models can be met more closely. It should be noted that this is an intensive method, involving the handling of specimens, so should be undertaken with caution in very vulnerable populations.

The identification software, I^3S, gave mixed results. It requires the dorsal wart pattern to be parallel to the camera lens, so the photography technique in the field must be precise. The reliability of it needs to be tested further, including whether inputter subjectivity affects it. Only then can it be either discounted or relied on more heavily in *E. calamita* studies. In terms of time, the study has found that I^3S can only be used if the number of photographs inputted is smaller than those to be compared.

In the future, the CR surveys should be continued with more surveys per year to gain better understanding of population fluctuations. All survey efforts must be comparable in terms of time taken, number of surveys and groups of surveyors.

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Research article

Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of Loons (*Gavia* spp.)

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ABSTRACT

Seabirds select suitable habitats at sea, but these habitats may be strongly impacted by marine spatial planning, including the construction of offshore wind farms (OWFs) and the associated ship traffic. Loons (*Gavia* spp.) are particularly vulnerable to anthropogenic activities and are also of high conservation status, making them particularly relevant to marine planning processes. We investigated the effects of OWF construction and ship traffic on Loon distributions in the German North Sea on a large spatial scale, using a 'before–after' control impact analysis approach and a long-term data set. Many OWFs were built in or close to core areas of Loon distributions. Loons showed significant shifts in their distribution in the 'after' period and subsequently aggregated between two OWF clusters, indicating the remaining suitable habitat. The decrease in Loon abundance became significant as far as about 16 km from the closest OWF. Ship traffic also had a significant negative impact on Loons, indicating that OWFs deterred Loons through the combined effect of ship traffic and the wind turbines themselves. This study provides the first analysis of the extensive effects of OWFs and ships on Loons on a large spatial scale. The results provide an essential baseline for future marine spatial planning processes in the German North Sea and elsewhere.

1. Introduction

Shallow-shelf sea areas have long been used by humans. The North Sea is amongst the most-intensively utilised sea areas worldwide for activities including fishing, transport, oil and gas drilling, and gravel extraction (Emeis et al., 2015; Halpern et al., 2008). The installation of offshore wind farms (OWFs) in many sea areas throughout Europe and elsewhere represents a relatively new human use requiring considerable attention in terms of the marine planning process. In order to meet their climate goals, many European governments have started to install and plan further OWFs within relatively large sea areas (e.g. Breton and Moe, 2009; Langston, 2010). Germany intends to extend its offshore power generation to 6,500 MW by 2020 and to 15,000 MW by 2030, leading to a large increase in the number of OWF sites, mainly in the German North Sea, making Germany one of the countries with the most extensive plans for OWF installations (Beiersdorf and Radecke, 2014). Seventeen OWFs are currently (2018) in operation, with five further ones under construction and several more being approved in German sea areas (BSH, 2017).

In terms of the process of marine spatial planning, these permanent installations at sea represent a major addition to other types of marine human activities, whilst competing with sea areas assigned for nature conservation (Emeis et al., 2015; Moksness et al., 2009; Nolte, 2010) and potentially overlapping with areas used by resting and foraging seabirds. Previous studies have pointed out contrasting effects (negative or positive) of OWFs on seabirds that vary strongly among areas and species (Dierschke et al., 2016; Drewitt and Langston, 2006; Fox and Petersen, 2006; Furness et al., 2013; Garthe and Hüppop, 2004; Masden et al., 2009). In addition, the construction and maintenance of OWFs is further associated with a strong increase in shipping activities in and around OWFs (Exo et al., 2003).

OWFs may have direct effects on birds such as collision of individuals with the turbines, with subsequent impacts on the whole population (Fox et al., 2006; Goodale and Milman, 2014; Masden et al., 2009). Furthermore, the energy budget and condition of individual birds may also be affected indirectly through the effects of OWFs on habitat loss and reduced food availability (Drewitt and Langston, 2006; Fox et al., 2006; Stienen et al., 2007), though the long-term effects of these indirect effects at the population level are hard to estimate (Fox et al., 2006; Goodale and Milman, 2014; Searle et al., 2017). However,

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birds have been shown to lose suitable resting and foraging habitats or to select less suitable sea areas (Stienen et al., 2007). Furthermore, they may need to increase their flight time by flying around OWFs on their way to suitable foraging sites (Drewitt and Langston, 2006; Masden et al., 2009). This study aimed to quantify the indirect effects (i.e. habitat loss by OWFs and associated ship traffic) on Loons (*Gavia* spp.) to provide baseline data for future studies that might address population consequences.

Loons belong to the most sensitive species group with respect to the avoidance of OWFs, as shown for single OWF sites in the North Sea (e.g. Dierschke et al., 2012, 2016; Leopold et al., 2010; Mendel et al., 2014; Petersen et al., 2006a, b; Welcker and Nehls, 2016). Furthermore, Red-throated Loons (*Gavia stellata*) are also very sensitive to ship traffic, demonstrating long flush distances in front of approaching vessels (Bellebaum et al., 2006) and significantly lower densities in areas with permanently higher ship traffic (Hüppop et al., 1994; Schwemmer et al., 2011). Their sensitive nature and the fact that a significant proportion of the biogeographic population occurs in European waters means that Loons are listed in Annex I of the EU Birds Directive and are considered to be particularly threatened with respect to human activities (e.g. Furness et al., 2013; Garthe and Hüppop, 2004). Negative effects on Loons at both the individual and population levels as a result of avoidance of OWFs cannot be ruled out (Dierschke et al., 2016, 2017), and Loons are therefore currently rated as a species group requiring particular consideration with respect to marine spatial planning in Germany and the UK (Busch et al., 2013).

Most Loons in the North Sea are Red-throated Loons (90%), with a minor proportion of Black-throated Loons (*G. arctica*; 10%) (Dierschke et al., 2012; Garthe et al., 2007). The German North Sea represents one of the most important resting sites for Loons with internationally important numbers, especially during spring migration (Garthe et al., 2007, 2015; Mendel et al., 2008; Skov et al., 1995), when around 20,200 Loons use German waters (Garthe et al., 2015). The 'Eastern German Bight' Special Protection Area (SPA) has been established to acknowledge the importance of this resting site and the high sensitivity of Loons with respect to human disturbances (Fig. 1). However, there is a potential conflict with the 'Butendiek' OWF, which was approved before but installed after the establishment of the SPA (Garthe et al., 2012), while further OWFs ('Helgoland Cluster') are located just south of the border of the SPA (Fig. 1).

Information on the long-term and large-scale effects of OWFs on Loons is currently limited and there has been no long-term comparison of their distributions before and after the installation of OWFs. Furthermore, the effects of increasing construction- and maintenance-related ship traffic have rarely been considered (Boon et al., 2010; Christensen et al., 2003).

We therefore hypothesized that Loons would avoid OWF areas and that their distribution patterns would differ before and after the installation of OWFs. We also hypothesized that the ship traffic associated with OWF sites would cause avoidance reactions among Loons. Against this background, this study aimed to shed light on five specific topics. (1) We had access to a long-term dataset covering the 14-year period before the installation of the OWFs ('before'). We therefore aimed to compare this information directly with the distribution of Loons after the installation of OWFs ('after'), using a long-term perspective not achievable in most previous studies. Mandatory operational monitoring of the four offshore windfarms in focus is still ongoing. (2) Most previous studies of the potential effects of OWFs on Loons have focussed on the effects of single OWF sites and their direct vicinities (see Dierschke et al., 2016). These therefore only allowed the reactions of Loons to be studied on a relatively small spatial scale, and could only show that Loon numbers were impacted within the respective site but could not show where they had moved to (Rexstad and Buckland, 2012). In contrast, the current study aimed to analyse the large-scale effects of multiple OWFs on Loon distribution, considering potential shifts between the 'before' and 'after' periods. (3) There is currently a need to

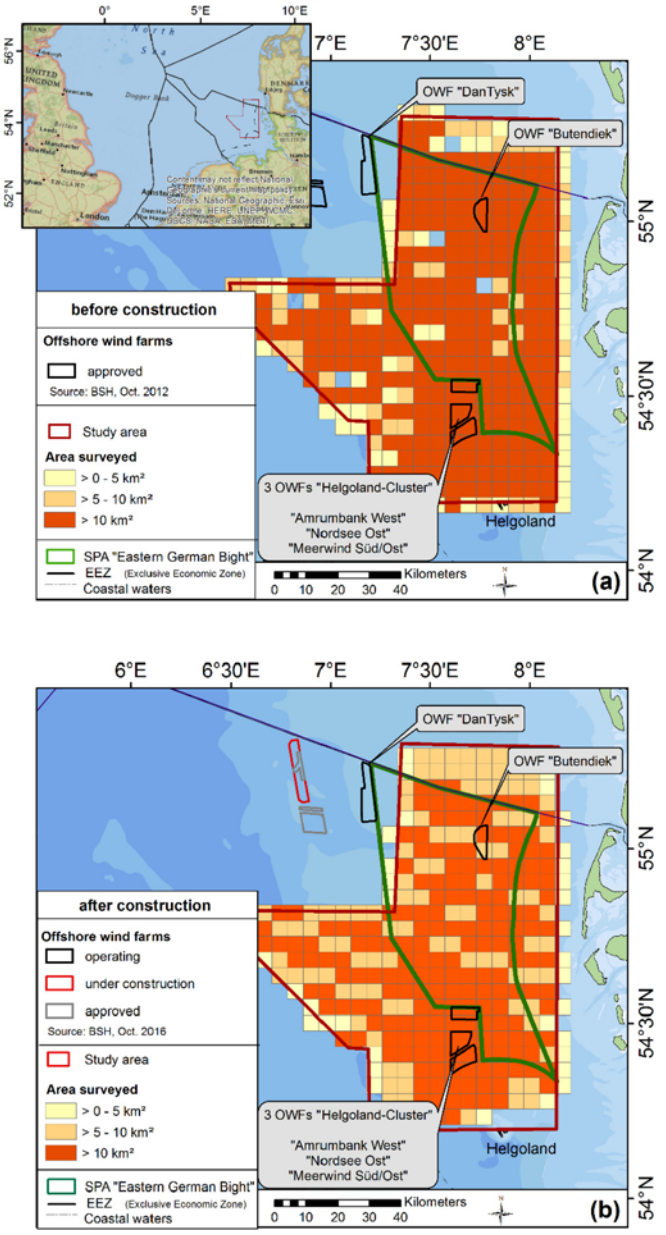


Fig. 1. Location of the study site within the south-eastern North Sea (inserted map in Fig. 1a) and in the eastern German Bight (North Sea) with locations of the different OWFs and the area surveyed for Loon abundance (yellow to red squares) across the 'Eastern German Bight' Special Protection Area (SPA; bold green line) for the 'before' (a) and 'after' periods of the analysis (b). Start of construction: 'Nordsee Ost' OWF during summer 2012; end of all construction works: 'Butendiek' OWF during summer 2015. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

disentangle the potential effects of OWFs from the effects of natural habitat characteristics that determine the distribution of Loons (Garthe, 1997; Winiarski et al., 2014). We therefore developed a model including stable natural parameters such as water depth and distance to land, as well as anthropogenic predictors such as distance to closest OWF and shipping traffic. (4) Given that the installation and maintenance of OWFs is associated with large increases in ship traffic, the effects of shipping need to be quantified and separated from the effects of the OWFs themselves. To date, this only has been analysed based on general ship densities (e.g. APEM, 2013; 2016; Leopold et al., 2014), while OWF ships present a dynamic source of disturbance for Loons.

This study therefore aimed to relate Loon and ship distributions at very high spatial and temporal scales by relating ship distributions derived from the Automatic Identification System (AIS) with Loon abundance assessed during aerial surveys. (5) Given a negative effect of OWFs on Loons, we aimed to quantify the avoidance distance to OWFs to draw conclusions about the degree of resulting (permanent) habitat loss.

In this study, we adopted two different approaches to analyse different aspects of the effects of OWFs on Loons: we used 'before' data to demonstrate the importance of the OWF areas before construction, and also focused on the simultaneous effects of OWFs and ships associated with OWFs after construction. The combined interpretation of these approaches allowed a comprehensive evaluation of the effects of OWFs on Loons.

2. Methods

2.1. Study area

The study was conducted within the eastern part of the Exclusive Economic Zone of the German North Sea, south of 55°17′ N, north of 54°11′, east of 6°30′ E, and west of 8°9′ E (Fig. 1a). The study site was located within an area 8–100 km off the Wadden Sea islands of northern Germany. The water depth ranged from 10 to 40 m. Loon distribution was recorded within the SPA 'Eastern German Bight' and beyond, and the study site therefore covered the core area of highest Loon densities within German waters (Garthe et al., 2015). The 'Butendiek' OWF is located in the core area of the SPA, while the 'Helgoland Cluster' OWFs are located at the border of the SPA and south of the core Loon distribution (Fig. 1a).

2.2. Recording Loon distribution and data processing

Loon distribution was recorded, both, in the period prior to OWF construction and in the period after construction:

(1) Before construction: These data cover the months of spring migration (i.e. March to April) of the years 2000–2013 and are the similar database as used by Garthe et al. (2015). The records originated from environmental impact assessment studies required for licensing procedures of offshore wind farms in the German EEZ and from seabird monitoring and research programmes (for details see Garthe et al., 2015; Fig. 1a). The data were recorded using visual aerial and ship-based surveys. Briefly, Loons were counted along transects of a known area, which allowed the densities to be computed (see Diederichs et al., 2002; Garthe et al., 2002 for a full description of both recording methods).
(2) After construction: These data also cover the months of spring migration (i.e. mainly March to April, but including the last week of February and the first week of May to enhance the sample size of surveys) of the years 2015–2017. Data originated from ongoing mandatory monitoring of the wind farms during operation, and from the 'Helbird' research project funded by the German Federal Ministry for Economic Affairs and Energy. Overall, data for the after period were based on 10 digital aerial surveys in 2015–2017 (Fig. 1b). Those data were obtained by video-based digital recordings instead of visual observations. Briefly, an aircraft sampled a transect of a known area using a video camera and all seabirds found were recorded and used to compute overall densities (for a detailed description of the method see Buckland et al., 2012; Thaxter and Burton, 2009). A change from visual to digital survey methods was mandatory for safety reasons because the flight altitude needed to be higher during the construction and operational phases of the turbines (168 m, instead of 91 m for visual observations), which excluded visual recordings.
(3) During construction: No data were considered in this study, as disturbance during the construction of the OWF is temporary and mainly associated with construction ships, and its contribution to the overall effect of the OWF on the Loon population was assumed to be of low importance in relation to the expected lifetime of the OWF (Christensen et al., 2003).

Visual observations of seabird distributions are known to underestimate birds in parts of the transect further from the observer (Buckland et al., 2001, 2015). We therefore applied a species-specific correction factor for aerial and ship-based observations, respectively (see Garthe et al., 2015 for details). However, no distance correction was necessary for the video-based digital surveys because the probability of detecting a bird was equal across the whole transect.

All three recording methods relied on the principle that transect sampling of birds could be used to compute densities. However, we did not compare absolute density values between the 'before' and 'after' periods, because the visual and digital methods have not been confirmed to produce the same absolute values (Buckland et al., 2012; Skov et al., 2016); this could only be tested by performing both methods at the same time, and no such dataset is currently available. Thus, both periods were compared by computing the relative deviance from the maximum density in each period in %, and using this to compare the distributions and locations of high-density areas of Loons between the two periods.

Data were spatially pooled in a grid with cells of 2.5 × 2.5 km for the 'before' and 'after' periods, for each of the three methods (visual aerial and ship-based surveys, video-based digital recordings), respectively. Bird numbers and monitored areas were each summed per grid cell, and eventually used to compute mean densities for each period, while geographical coordinates were averaged for each cell.

2.3. Integrating covariates for the 'before–after' control impact (BACI) approach

We related the average distribution data for Loons with environmental variables using ArcGIS (version 10.3; Environmental System Research Institute, 2016). The environmental variables included: (1) *dist_coast* = minimum distance to the mainland and larger islands (except Helgoland); (2) *dist_helgoland* = minimum distance to Helgoland; (3) *dist_owf* = minimum distance to the border of the OWF; and (4) *mean_depth* = mean water depth.

This first model, hereafter named the BACI approach, did not consider the effect of ships because ship data at a sufficiently high spatio-temporal resolution were only available for the 'after' period. To distinguish between the effect of the OWFs and the effect of ship traffic on Loons, we therefore developed a second model (ship model) using only the data from the 'after' period.

To merge the environmental variables with the bird-count data in an optimal way, we first pooled the covariates to a spatial grid of 2.5 × 2.5 km, and then fitted each covariate with a generalised additive model (GAM) using the function gam() in the R-package mgcv (R Core Team, 2017; R version 3.4.2; Wood, 2006). We used only latitude and longitude as a smooth 2D-predictor based on cubic splines with the maximal degree of freedom, so that the result represents a cubing interpolation on the given (possibly irregular) grid. Thirdly, we used the predict() function to predict the values straight to the coordinates as given in the pooled bird-count data. Finally, the additional categorical variable *owf_zone* for 'inside OWF-affected area' vs. 'outside OWF-affected area' was defined for two different zones: 1) inside: ≤ 3 km vs. outside: > 3 km (measured from the nearest turbine), given that OWF-associated ships operate mainly within a 3 km radius around the OWF and this distance class has been used in previous studies of the impact of single OWFs (Vanermen et al., 2015a; Welcker and Nehls, 2016); and 2) inside: ≤ 10 km vs. outside: > 10 km, because an initial analysis showed the greatest decrease in Loon densities up to a distance of 10 km from the turbines.

2.4. Set up and validation of regression models for the BACI approach

The BACI approach is based on surveying a potentially impacted situation and a control situation before the impact (variable '*period*'), and relative comparisons of spatial and temporal differences can then be used to extract the unbiased impact (Schwarz, 2014; Smith, 2002). We formulated the BACI approach within the framework of generalised additive mixed models (GAMMs), which are known to describe biological count data appropriately (Zuur et al., 2007, 2009; 2012). We used a continuous linear or smooth predictor measuring the distance to the border of the next OWF. This allowed us to estimate how the abundance of Loons changed in relation to the distance from the OWF and to estimate avoidance distances. Notably, we introduced a variable for the observation method ('visual ship-based surveys' vs. 'visual aerial surveys' vs. 'digital aerial surveys') as a random intercept to account for differences in detection among these methods. We were aware that this variable was partially collinear with the variable 'period' because only digital aerial surveys were used 'after' and only visual surveys were performed 'before'. Importantly, the estimation of the interaction term 'period x wind_farm' (see below) representing the BACI approach was not influenced by this, because only relative differences in Loon densities were evaluated.

This approach produced the following full model for the BACI approach (not yet thinned regarding its predictors; see below):

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f}(\mathrm{mean_depth_j}) + \mathrm{f}(\mathrm{dist_coast_j}) + \mathrm{f}(\mathrm{dist_helgoland_j}) + \mathrm{s}(\mathrm{latitude,longitude}) + [\mathrm{wind\ farm_j}] + \mathrm{period_j} + [\mathrm{wind\ farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (1)$$

where $\varepsilon_{ij} \sim N(0, \sigma^2)$ and $u_i \sim N(0, \sigma_u^2)$ were independent and identically distributed. Here, y_{ij} is the vector of bird numbers, where the index j refers to the observation number and i is related to the method-ID. $f()$ depicts either a linear term or a cubic regression spline $s()$ (tested during predictor selection), where, in the case of a spline, the optimal number of knots was estimated via cross-validation. The variable [*wind_farm*$_j$] was either considered as a linear term, *dist_owf*$_j$ measuring the distance to the next wind turbine, as an additive smoother, *s*(*dist_owf*$_j$), or as a bivariate variable, *owf_zone*$_j$, the latter distinguishing between 'inside OWF-affected area' and 'outside OWF-affected area'. For each model, an appropriate probability distribution was selected for y_{ij} via Akaike Information Criterion (AIC; Akaike, 1973) analysis (see below).

We modified the common selection and validation strategies to validate the optimal GAMM model (Field et al., 2012; Korner-Nievergelt et al., 2015; Zuur, 2012; Zuur et al., 2009, 2010; 2012) using the following steps: (1) Based on the entire model (1), we selected an appropriate probability distribution/stochastic part of the model using the AIC. Namely, we compared Poisson-, negative binomial-, Tweedie-, zero-inflated Poisson distribution, and observation-level random intercept Poisson models. All five probability distributions are known to describe the stochastic part in regression models of (overdispersed) count data reasonably well (Kokonendji et al., 2004; Korner-Nievergelt et al., 2015; Linden and Maentyniemi, 2011; Zuur et al., 2012). (2) The optimal model regarding the set of fixed-effect predictors was selected from the full model by comparing 16 different models. (3) Model validation was carried out by visual inspection of the residual plots to assess all the required model assumptions (Zuur et al., 2010). Corresponding auto-correlation structures were added to the model if required.

AIC favoured a negative-binomial distribution, and subsequent predictor selection produced the following final model:

$$\log(y_{ij}) = \beta_0 + u_i + \beta_1 \mathrm{dist_coast_j} + \mathrm{s}(\mathrm{latitude,longitude}) + [\mathrm{wind_farm_j}] + \mathrm{period_j} + [\mathrm{wind_farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (2)$$

Residual analysis revealed no violation of linearity, homogeneity, independence, or normality of the random intercept.

2.5. Integrating covariates for the ship model

Ship traffic has been shown to have a significant effect on Loon distribution (Bellebaum et al., 2006; Schwemmer et al., 2011), and ship traffic in the study area has increased greatly due to the construction and maintenance of OWFs. It is therefore important to disentangle the effects of these two sources of anthropogenic activities (OWFs and ship traffic) on Loons. Ship traffic shows temporal inhomogeneity, with more traffic in the morning and evening hours, and it was therefore necessary to consider the data spatio-temporally instead of purely spatially, as with the BACI approach. Data were only used for five digital-survey flights from the 'after' period because no real-time ship data were available for the 'before' period or for any other survey days during the 'after' period. Bird data were spatially assigned to an optimal grid of 2.5 × 2.5 km for each survey day separately and treated as described above. To consider the time, we also calculated the mean time at which the Loon observations were recorded for each grid cell.

Data on ship distributions were recorded in parallel with the digital-survey flights to record Loon distribution using an AIS spotter (www.aisspotter.com). Because the ship data consisted of irregular position data in terms of time and space, they were linearly interpolated to obtain positions at least every minute. To merge the ship data with the Loon-distribution data, it was assumed that all ships within the time interval $[t - \delta_t, t]$ and within a circle around (x, y) with radius r may influence bird density, for each time point t and each pair of spatial coordinates (x, y). Given that the optimal values δ_t and r are not known *a priori*, we tested all existing combinations between $\delta_t \in \{2, 60, 120, 180, 250, 300, 350, 400, 600, \infty\}$ sec and $r \in \{1, 2, 3, 4, 5, 6, 7, 8, 9, 10\}$ km, and created a separate variable counting all ships within the given time and space interval for each of the 100 combinations. Here, $\delta_t = \infty$ depicts a case where all available ship data have only been spatially correlated to bird-count data without considering temporal distance to the observations. We subsequently compared 100 resulting regression models (see below) to find the optimal values of δ_t and r. However, the AIC value was not appropriate for selecting the optimal model because the ship densities and OWF-related variables were collinear, and the model with only one of both variable types would be favoured due to the parsimony of the AIC-based selection. In contrast, we aimed to consider both (collinear) variables to distinguish explicitly between the unique effects of ships and wind turbines on Loon abundance. An appropriate measure should thus relate the effect size of the ship-dependent variable with its reliability. Hence, we selected the model with the highest $|\beta|/SE_\beta$ value, where β is the ship-related regression coefficient and SE_β is its standard error.

2.6. Set up and validation of regression models for the ship model

The GAMMs were set-up as described above for the BACI approach. Notably, the ID of the digital-survey flight was introduced as a random intercept to account for different numbers of birds or different monitoring conditions between surveys.

This produced the following GAMM structure of the ship model (not yet thinned regarding its predictors):

$$\log(y_{ij})\beta_0 + u_i + \mathrm{f}(\mathrm{mean_depth_j}) + \mathrm{f}(\mathrm{dist_coast_j}) + \mathrm{f}(\mathrm{dist_helgoland_j}) + [\mathrm{wind_farm_j}] + [\mathrm{ship_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (3)$$

where $\varepsilon_{ij} \sim N(0, \sigma^2)$ and $u_i \sim N(0, \sigma_u^2)$ were independent and identically distributed. Here, y_{ij} is the vector of bird numbers, where the index j refers to the observation number and i is related to the survey flight ID. $f()$ depicts a linear or smooth predictor (tested during AIC-based predictor selection). The variable [*wind_farm*$_j$] was either considered as a binomial predictor ('inside' vs. 'outside'), a linear term (distance to the

OWF border), or a cubic regression spline depending on the latter. The variable [$ship_number_j$] was considered as the total number of temporally and spatially related ships, additionally depending on the *a priori* defined parameters δ_t and r (see above). In contrast to the BACI approach, we did not consider a spatial smooth because this predictor would interfere with the correct estimation of [$wind_farm_j$]. GAMM-model selection and validation strategies were performed as described for the BACI approach (see above), including integration of the appropriate autoregression structures (if required).

AIC-based selection of the probability distribution again favoured a negative-binomial distribution. The optimal values of δ_t and r required to blend the observation and ship data showed that the highest (β/SE)-values (indicating high precision of the ship-related regression coefficient) were $\delta_t = 5$ min and $r = 5$ km. Subsequent predictor selection revealed the following final model:

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f}(\mathrm{mean_depth_j}) + \beta_1 \mathrm{dist_coast_j} + \mathrm{s}(\mathrm{dist_helgoland_j}) + [\mathrm{wind_farm_j}] + [\mathrm{ship_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (4)$$

where $s()$ depicts the cubic regression splines with optimal degrees of freedom estimated via cross-validation.

Analysis using different sizes of the underlying spatial grid for spatio-temporal pooling revealed an optimal grid size of 2.5 × 2.5 km, leading to a temporal autocorrelation of model residuals of order 2 (in contrast to the model based on raw data, where the autoregressive order (AR order) was > 30). Model-validation plots indicated no violation of linearity or homogeneity, spatial residual plots and a semi-variogram indicated no violation of spatial independence, and a plot of the partial autocorrelation function (pACF-plot) revealed a temporal autocorrelation of approximately order 2, which was integrated as an AR(2)-structure into the model.

3. Results

3.1. Loon abundance before and after OWF installation

The spatial distribution patterns of Loons changed profoundly between the 'before' and 'after periods (Fig. 2). During the 'before' period, the core area with the highest Loon densities clearly overlapped the area of the planned 'Butendiek' wind farm, while moderately high densities stretched out to the area of the planned 'Helgoland Cluster'. In contrast, there was a clear shift to the area located between these two OWF sites during the 'after' period (Fig. 2). The areas of the OWFs themselves, as well as the immediate vicinities, showed extremely low abundances of Loons during the 'after' period. The core area of Loons during the 'after' period was thus still located in the centre of the SPA, but the birds were more aggregated within the still-undisturbed sea area.

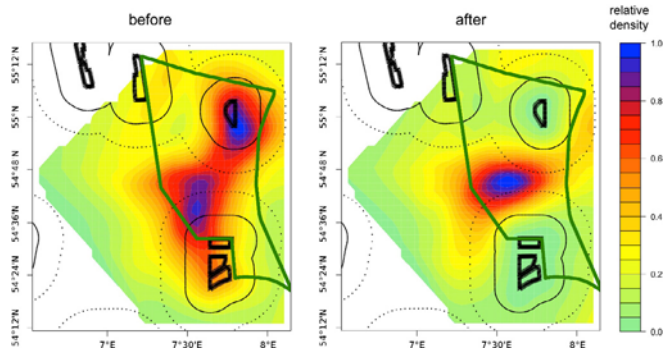


Fig. 2. Spatial density plots of predicted Loon distributions 'before' vs. 'after' the construction of OWFs, based on the BACI-GAMM. Bold black lines: OWFs; thin black lines: 10 km distance buffer; dotted black lines: 20 km distance buffer; bold green line: Special Protection Area.

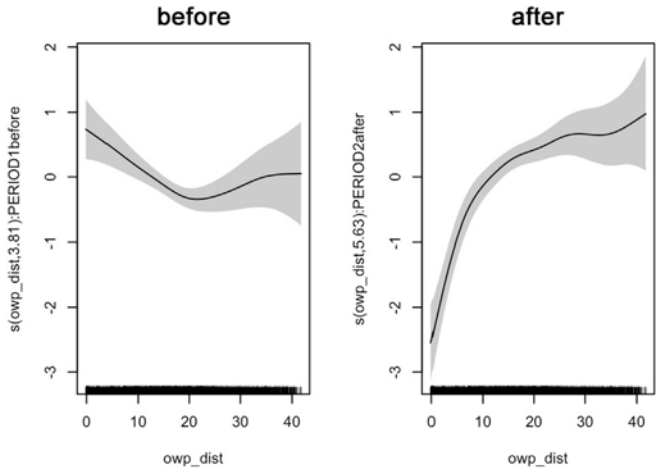


Fig. 3. Change in Loon abundance in relation to distance from the closest OWF site 'before' (left) and 'after' construction of the OWFs. Smoothed curve: predicted number of Loons at a given distance from the closest OWF; shaded area: 95% confidence interval; small lines on the x-axis: observations of Loons at a given distance from the OWF.

We also introduced the distance from the wind farm as a smooth term, estimated separately for each period. This revealed a striking difference between the two periods (Fig. 3): the 'before' plot suggested that the future wind farm areas were sites with naturally increased Loon abundance, while the 'after' plot showed a strong decline in Loon abundance due to the OWFs (Fig. 3). The start of this decline was already visible at > 20 km from the OWFs (see also dotted black lines in the 'after'-plot in Fig. 2b). To determine the distance from the wind farm at which the decline in abundance was significant, we approximated the first derivative of the corresponding smooth (Fig. 3 'after') by calculating its first finite difference.

To determine the distance at which the change in Loon density became significant, we calculated confidence intervals for the first derivatives via bootstrap analysis and subsequently evaluated where the lower confidence interval intersected with zero. This occurred at around 16.5 km from the OWFs (Fig. 4). However, the greatest decline in density was at distances within 10 km from the OWF (Figs. 3 and 4). Avoidance of wind farms within 10 km was also clearly visible in the distribution maps (solid black lines in Fig. 2b).

Additionally, the binomial wind farm-related variable *owf_zone* was highly significant for both radii (3 or 10 km, respectively). The abundance of Loons decreased highly significantly by 94.5% inside the 3 km zone around the OWFs within the study site (interaction term in Table 1; $\beta = -2.9$, $p < 0.001$), while the abundance was still decreased by 83.7% inside the 10 km zone (Table 2, $\beta = -1.8$, $p < 0.001$). The distance to land (*dist_coast*) had no significant effect on Loon densities (Table 1; Table 2).

3.2. Distinguishing between effects of ships and OWFs

Loon densities were still reduced if ships were included in the overall model as a predictor for the 'after' period, as was the case without considering the effect of ships, as shown above. Applying a 3 km radius around the wind farms, OWFs alone reduced the Loon density by 70.8% compared with the sea areas outside the OWFs ($p < 0.001$; Table 3). If the radius was extended to 10 km around the OWFs, the Loon density was still reduced by 44.5% ($p < 0.001$) by the OWFs alone.

When ships as single predictor were removed from the model, the estimated effect of OWFs (now combined with the effect of the ships) on Loons was 84% using a 3 km radius ($p < 0.001$). This suggested that

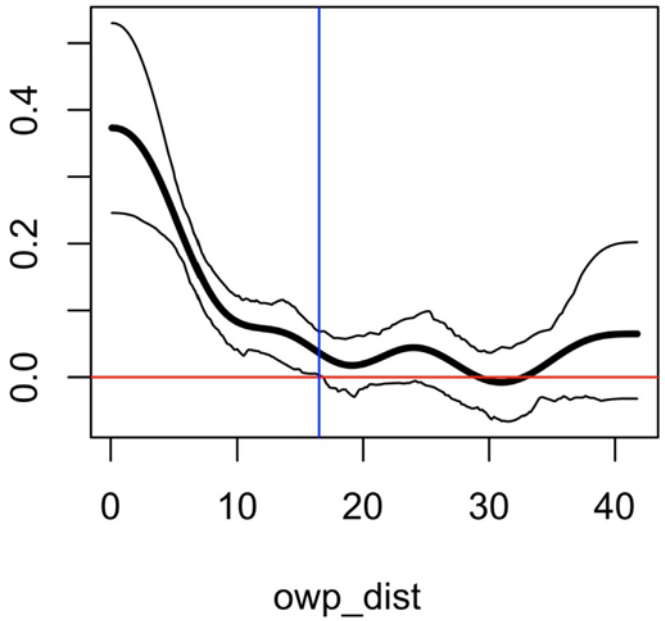


Fig. 4. First finite difference of the smooth depending on the distance from the closest OWF, partially evaluated for the 'after' period. Red line indicates a derivative of zero, blue line indicates distance at which the derivative was significant. Thick black line corresponds to the first derivative; thin black lines depict 95% confidence intervals.

Table 1
Regression results of the BACI approach–GAMM using the binomial variable 'inside wind farm' vs. 'outside wind farm' (*owf_zone*) for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	−1.05	1.17	−0.90	0.37
period[after]	0.40	0.85	0.47	0.64
owf_zone[a][inside]	0.70	0.13	5.07	< 0.001
dist_coast[b]	0.02	0.02	0.70	0.43
period[after]xowf_zone[inside]	−2.90	0.22	−13.16	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.

Table 2
Regression results of the BACI approach–GAMM using the binomial variable 'inside wind farm' vs. 'outside wind farm' (*owf_zone*) for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	−1.41	1.17	−1.21	0.23
period[after]	0.73	0.95	0.76	0.45
owf_zone[a][inside]	0.66	0.12	5.59	< 0.001
dist_coast[b]	0.02	0.02	1.01	0.31
period[after] xowf_zone[inside]	−1.81	0.12	−15.26	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.

Table 3
Regression results for the *ship-owf*-approach–GAMM distinguishing between the effect of ships and the effect of OWFs in the 'after' period for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	0.29	0.58	0.5	0.62
owf_zone[a][inside]	−1.23	0.31	−4.03	< 0.001
dist_coast[b]	−0.01	0.01	−0.55	0.58
n_ships[c]	−0.37	0.08	−4.82	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.
[c] Number of ships.

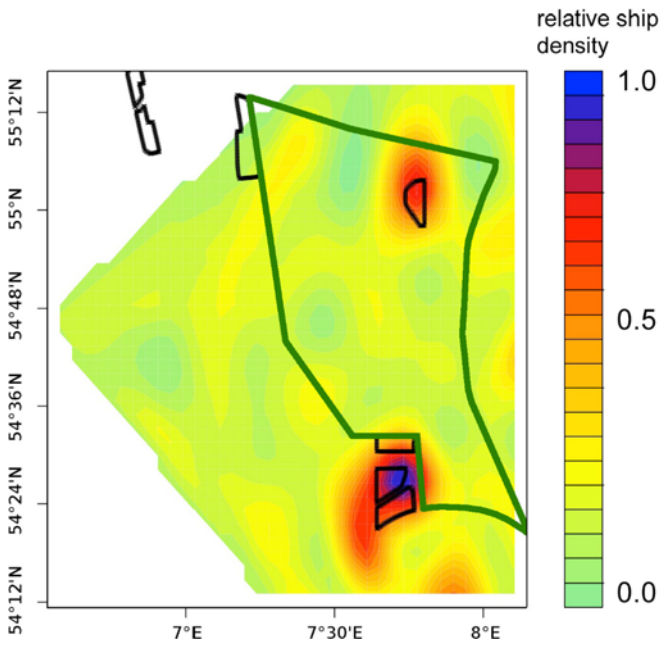


Fig. 5. Spatial density plot of ship distribution in the 'after' period based on AIS data.

ships also had a strong negative effect on Loon abundance, accounting for at least 14% of the joint OWF–ship effect.

Thus, in the ship model, the effect of OWFs alone was not as strong as estimated by the BACI approach (i.e. without considering ship traffic; > 94% and > 84%, respectively). There are two possible explanations for these different estimations. (1) the ship model was only fitted using data from the 'after' period because no ship data were available for the 'before' period. Hence, the estimated reduction in effect does not take account of the fact that bird densities within the OWFs showed the highest Loon abundances before the construction of the farms (see above), leading to a strong underestimation of the reduction effect. (2) Although the ship model considered the effect of ships, these were at least partially correlated with OWF location (Fig. 5). Thus the BACI approach actually estimated the joint reduction effect of OWFs and ships, whereas the ship model evaluated both impacts separately, which may have led to a reduction in the OWF effect compared with the BACI approach.

Indeed, the ship model showed a significant negative impact of ships on Loon abundance (Tables 3–4), with a highly significant decline of 31% in abundance for each additional ship in the spatio-temporal range of the Loons (i.e. 5 min and 5 km from the Loon sighting; see Methods) ($p < 0.001$). This suggests that one in three Loons left the area as one ship approached. The spatial component of ship disturbance was much stronger than the temporal component; i.e. our regression models selecting for the optimal δ_t and r revealed that ships within 5 km had a strong impact on Loon abundance, whereas the time lag between

Table 4
Regression results for the *ship-owf*-approach–GAMM distinguishing between the effect of ships and the effect of OWFs in the 'after' period for a radius of 10 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	0.73	0.58	1.26	0.20
owf_zone[a][inside]	−0.59	0.17	−3.51	< 0.001
dist_coast[b]	−0.01	0.01	−1.00	0.32
n_ships[c]	−0.48	0.07	−6.44	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.
[c] Number of ships.

the Loon sighting and the AIS signal of the ship was less relevant (with an optimum at approx. 5 min). This suggests that ships may affect Loons most strongly at a distance of ≤5 km.

As seen with the BACI approach, the distance to land had no significant influence on Loon abundance (Tables 3–4).

4. Discussion

4.1. Distribution patterns before and after OWF installation

Our results demonstrated that the distribution patterns of Loons, which had remained stable over a period of many years (Garthe et al., 2015), were substantially altered at both small and large spatial scales by the installation of OWFs in the German North Sea. We developed our BACI approach on a solid database including 14 years of large-scale surveys in the period 'before' OWF installation. To the best of our knowledge, all previous reports have been based on a maximum of 1–3 years of data prior to the construction of OWFs, and have mostly focused on the effect of a single OWF (e.g. Leopold et al., 2013; Petersen et al., 2014). Although we were unable to compute absolute differences in Loon populations between the two periods due to a change in survey methods, our results demonstrated profound large-scale shifts in distribution patterns, as well as significant avoidance of the OWF areas.

We observed a shift in the Loon-abundance hotspot to the western–central area of the SPA that remained undisturbed by OWFs in the 'after' period. This hotspot is located about 20 km distant from all surrounding OWFs. Several previous studies have highlighted the environmental parameters that are most important for determining Loon distribution patterns. Frontal systems are expected to increase prey availability for Loons (Skov and Prins, 2001), while nearshore and shallow sandy sea areas also play a major role (O'Brien et al., 2008; Skov and Prins, 2001; Skov et al., 2016). Our results suggest that the area of the 'Butendiek' OWF, which was installed in the northern part of the SPA, was of particular importance for Loons before the construction of this OWF, given that this was the area of maximum Loon abundance during the 'before' period. The 'Helgoland-Cluster' OWFs are located south-west of the border of the SPA, and our results showed that, in contrast to the 'Butendiek' area, Loon abundances in the 'before' period were significantly lower compared with abundances within the SPA. However, Loons are known to occur here regularly (Garthe et al., 2015).

One aim of this study was to disentangle the importances of natural habitat structures and anthropogenic pressures on Loons. Our modelling approach showed that natural habitat predictors, such as distance to the coast/Helgoland and water depth did not play major roles compared with the effects of OWFs and shipping (see below). This suggests that anthropogenic pressures are the most important factors driving the distribution patterns of Loons within their natural hot spots.

Still, we cannot completely rule out that undetected changes in ecological conditions might have additionally led to the shift in distribution patterns. For instance, it could be assumed that Loons might have followed shifts in their prey community. However, given that Loons are known to feed on a variety of fish species (Guse et al., 2009), a shift in fish distribution that could account for the change in Loon distribution seems highly unlikely. The reef effect is even known to likely increase benthic and fish communities inside OWFs (e.g. Vandendriessche et al., 2015; Vanermen et al., 2015a) which in turn may enhance the quality of these sites for piscivorous seabirds. However, given that these sites were avoided by Loons, despite of a likely higher fish availability and as Loon distribution patterns had been stable over a period of many years in the 'before' period (Garthe et al., 2015), it seems to be convincing that OWFs and associated ship traffic are the main factors explaining the shifts in distribution patterns.

Incorporating distance from the nearest OWF as a smoothed term in the model allowed us to highlight the fact that Loons reacted as far as 20 km from OWFs, with significant changes in densities at a distance of 16.5 km and the greatest changes in abundance within 10 km. These values were higher than those reported in previous studies (summarized in Dierschke et al., 2016; Welcker and Nehls, 2016). However, most previous studies only investigated local avoidance effects (often only up to 4 km distance; Leopold et al., 2013; Petersen and Fox, 2007, Petersen et al., 2006a,b; Welcker and Nehls, 2016) and were therefore unable to detect any larger-scale avoidance reactions. This highlights the importance of a sufficiently large-scale approach and the inclusion of multiple OWF sites (Rexstad and Buckland, 2012), as in the current study. To emphasize the importance of scale, we quantified the effects of OWFs on Loons by defining the affected sea areas by both 3 km and 10 km radii.

The 3 km distance class was chosen based on previous studies that showed avoidance distances for single OWFs up to this value (Vanermen et al., 2015a; Welcker and Nehls, 2016). However, our results suggest that this distance was too short, based on the effects of multiple OWFs on a larger spatial scale.

The reason for the relatively large-scale effect of OWFs on Loons detected in the current study is not completely clear. It is possible that visual cues are not the only reason for the large disturbance distance. Previous studies showed that OWFs not only affected seabirds and other marine wildlife directly (Bergström et al., 2014; Goodale and Milman, 2014; Lindeboom et al., 2011), but may additionally cause changes in the abiotic environment, such as sediment properties and water stratification due to turbulence caused by the piles (Carpenter et al., 2016; Nagel et al., 2018). Carpenter et al. (2016) pointed out that an individual OWF may enhance mixing of the water column, with a cascade of effects on the whole ecosystem in an area of 10–20 km from the OWF, though the physical–biological interactions remain unclear. This was in accordance with the disturbance distance of Loons found in the current study. Petersen et al. (2014) also showed significantly lower Loon abundances up to 13 km from OWFs, which also matched the results of the current larger-scale approach.

Finally, it is important to critically explore the question of the power of the data used in this study. For the type of data used, previous investigations have shown that high survey intensities are required to safely trace declines in seabird populations, mainly as a result of high variability in distribution patterns (e.g. MacLean et al., 2013; Vanermen et al., 2015b). However, compared to our study that was conducted over a large sea area, both studies mentioned above focussed on rather small study sites, likely enhancing small-scale variability in counting data. According to Vanermen et al. (2015b) the statistical power after 10 years of survey was sufficiently high to detect reliable changes. For the 'before' period, 13 years of data were available for our BACI approach, indicating a valid data base. In contrast, the 'after' period only consists of 10 aerial surveys over a period of three years, suggesting that the data base for the 'after' period may still be too weak. However, the significant negative and consistent effects of OWFs and associated ship traffic on Loon distribution during all surveys of the 'after' period indicates that the data base is sufficient to yield valid results. Nevertheless, it will be necessary to enhance the data base for the 'after' period by future surveys to confirm the results and to enhance the statistical power.

4.2. Distinguishing between the effect of ships and OWFs

The installation of OWFs causes a substantial increase in ship traffic in the surrounding area due to maintenance and service activities (Exo et al., 2003). Although ship traffic is known to affect the distribution patterns of seabirds and particularly of Loons (Bellebaum et al., 2006; Schwemmer et al., 2011), the combined effect of OWFs and their associated ship traffic has rarely been reported; however, the few available studies noted a significant impact of ship traffic on Loon distribution (APEM, 2013, 2016; Leopold et al., 2014; Skov et al., 2016). Loons have been shown to exhibit a behavioural response to approaching ships, and flight distances of up to 2 km have been

documented (Bellebaum et al., 2006; Schwemmer et al., 2011). This corresponds to the current results, which suggested a significant reduction in Loon densities within a radius of up to 5 km from the vicinity of ships, with the temporal aspect of ship distribution having little effect.

Inclusion of ship abundance in the model showed a reduced density of Loons of up to 70% based on the 3 km distance zone. This reduction could be considered to reflect the effect of the OWFs alone. In contrast, the joint effect of OWFs and ships led to a reduction of 84%, indicating the additional negative impact of ships on Loon densities. The exact reduction in densities due to ships alone could not be computed reliably because of the collinearity of ship traffic and OWFs. Importantly, their mobile nature means that ships are both spatially and temporally variable predictors, and a reliable estimation of their overall effects on birds will always be biased. This issue will remain difficult to address even in future studies, given that ships aggregate strongly in the vicinity of OWFs and present no fixed predictor.

The greater reduction in Loon densities following inclusion of ship traffic in the model demonstrates the importance of reviewing the cumulative impact of multiple anthropogenic pressures in the marine environment. Previous studies have focussed on cumulative effects simply by investigating the combined effects of multiple OWFs (Busch et al., 2013; Desholm, 2009; Dierschke et al., 2003, 2006, Fox et al., 2006; King et al., 2009; Mendel and Garthe, 2010). However, given the strong effect of ships on Loon abundance, it seems necessary to include other anthropogenic pressures in estimates of cumulative effects on Loon abundance in general.

4.3. Conclusions

The large-scale avoidance effects of OWFs (and ships) on Loons suggest that Loons are unlikely to suffer from enhanced direct mortality, e.g. because of collisions (Leopold et al., 2010; Petersen et al., 2006a,b; this study). Furthermore, a low flight altitude of only up to 10 m above the sea surface (Van Bemmelen et al., 2011) reduces the collision risk for Loons. Indirect effects, such as habitat loss, are thus likely to be key factors affecting Loons in relation to OWFs. However, the consequences of such indirect effects e.g. on population levels of seabirds, and density-dependent effects are hard to assess, and appropriate methodologies are largely lacking (Green et al., 2016; Horswill et al., 2017). When assessing the consequences of habitat loss due to the installation of OWFs and the associated enhancements in ship traffic, it is essential to consider which alternative sea areas could be used as resting and foraging grounds. In the current case, alternative sites seemed to be very limited because the SPA was virtually surrounded by OWFs. This might explain why Loons tended to concentrate in the centre of the SPA rather than moving outside it.

Although it was not possible to compute absolute differences in abundance between the 'before' and 'after' periods in this study, it is hoped that this issue will be resolved when enough data become available from parallel digital and visual surveys of sea areas where visual observations are still allowed. However, the relative reduced densities of Loons with respect to OWFs and ship traffic as well as the avoidance distances provided in the current study will serve as a baseline for further studies. A suitable approach for quantifying the overall habitat loss for Loons would involve computing the relative proportion of habitat loss within a certain area (e.g. within the SPA). Dierschke et al. (2006) suggested summing the total OWF areas and adding an additional buffer zone to assess the overall habitat loss. Applying this approach to the current study allowed the minimum habitat loss due to the OWFs in the SPA to be computed, indicating that complete loss of the sea area within a 3 km radius around the OWFs for Loons (as strongly supported by the current study) would equate to a loss of 8.8% of the SPA (overall size 3,135 km^2) for Loons. This should be regarded as an absolute minimum, given that our results clearly showed that the density of Loons was greatly reduced beyond 3 km from the nearest OWF.

Although we are not able to compare absolute density values between the 'before' and 'after' periods, our results indicated that Loons aggregated in the centre of the SPA after OWF installation, representing an increase in Loon density in a much smaller sea area. Given that Loons tend to occur in comparatively small flocks, only occasionally exceeding 5–10 individuals/km^2 (Garthe et al., 2015; O'Brien et al., 2012), this change in distribution might promote density-dependent effects (Blanc et al., 2006; Horswill et al., 2017; Lewis et al., 2001). A possible shift towards suboptimal habitats may lead to suboptimal body conditions prior to breeding, which could in turn reduce the reproductive success and enhance mortality in adult birds (Coulson et al., 1983; Hüppop, 1995). Even a slight increase in the mortality of adult Loons of only 0.3% can have significant negative effects on population levels (Rebke, 2005).

To assess the role of habitat loss on Loons, it is crucial to know if habituation to OWFs will occur or if the habitat loss will be permanent. Although studies from the UK and The Netherlands have indicated slight (though insignificant) increases in Loon abundances after 4–5 years since construction, studies from Denmark have shown no signs of habituation (Petersen and Fox, 2007; Petersen et al., 2008). Similarly, the current study found no habituation 3 years after construction. However, the monitoring of the operating wind farms is still ongoing and thus results on habituation are preliminary. Given that the degree of habituation remains very unclear, we strongly recommend the need for long-term monitoring to assess any potential large-scale effects of cumulative anthropogenic drivers on Loon distribution, particularly within the most relevant sea areas for Loons (e.g. Vanermen et al., 2015a,b).

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Ministry of Housing,
Communities &
Local Government

National Planning Policy Framework

February 2019
Ministry of Housing, Communities and Local Government

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15. Conserving and enhancing the natural environment

170. Planning policies and decisions should contribute to and enhance the natural and local environment by:

a) protecting and enhancing valued landscapes, sites of biodiversity or geological value and soils (in a manner commensurate with their statutory status or identified quality in the development plan);

b) recognising the intrinsic character and beauty of the countryside, and the wider benefits from natural capital and ecosystem services – including the economic and other benefits of the best and most versatile agricultural land, and of trees and woodland;

c) maintaining the character of the undeveloped coast, while improving public access to it where appropriate;

d) minimising impacts on and providing net gains for biodiversity, including by establishing coherent ecological networks that are more resilient to current and future pressures;

e) preventing new and existing development from contributing to, being put at unacceptable risk from, or being adversely affected by, unacceptable levels of soil, air, water or noise pollution or land instability. Development should, wherever possible, help to improve local environmental conditions such as air and water quality, taking into account relevant information such as river basin management plans; and

f) remediating and mitigating despoiled, degraded, derelict, contaminated and unstable land, where appropriate.

171. Plans should: distinguish between the hierarchy of international, national and locally designated sites; allocate land with the least environmental or amenity value, where consistent with other policies in this Framework[53]; take a strategic approach to maintaining and enhancing networks of habitats and green infrastructure; and plan for the enhancement of natural capital at a catchment or landscape scale across local authority boundaries.

172. Great weight should be given to conserving and enhancing landscape and scenic beauty in National Parks, the Broads and Areas of Outstanding Natural Beauty, which have the highest status of protection in relation to these issues. The conservation and enhancement of wildlife and cultural heritage are also important considerations in these areas, and should be given great weight in National Parks

[53] Where significant development of agricultural land is demonstrated to be necessary, areas of poorer quality land should be preferred to those of a higher quality.

and the Broads[54]. The scale and extent of development within these designated areas should be limited. Planning permission should be refused for major development[55] other than in exceptional circumstances, and where it can be demonstrated that the development is in the public interest. Consideration of such applications should include an assessment of:

a) the need for the development, including in terms of any national considerations, and the impact of permitting it, or refusing it, upon the local economy;

b) the cost of, and scope for, developing outside the designated area, or meeting the need for it in some other way; and

c) any detrimental effect on the environment, the landscape and recreational opportunities, and the extent to which that could be moderated.

173. Within areas defined as Heritage Coast (and that do not already fall within one of the designated areas mentioned in paragraph 172), planning policies and decisions should be consistent with the special character of the area and the importance of its conservation. Major development within a Heritage Coast is unlikely to be appropriate, unless it is compatible with its special character.

Habitats and biodiversity

174. To protect and enhance biodiversity and geodiversity, plans should:

a) Identify, map and safeguard components of local wildlife-rich habitats and wider ecological networks, including the hierarchy of international, national and locally designated sites of importance for biodiversity[56]; wildlife corridors and stepping stones that connect them; and areas identified by national and local partnerships for habitat management, enhancement, restoration or creation[57]; and

b) promote the conservation, restoration and enhancement of priority habitats, ecological networks and the protection and recovery of priority species; and identify and pursue opportunities for securing measurable net gains for biodiversity.

175. When determining planning applications, local planning authorities should apply the following principles:

a) if significant harm to biodiversity resulting from a development cannot be avoided (through locating on an alternative site with less harmful impacts),

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[54] *English National Parks and the Broads: UK Government Vision and Circular 2010* provides further guidance and information about their statutory purposes, management and other matters.
[55] For the purposes of paragraphs 172 and 173, whether a proposal is 'major development' is a matter for the decision maker, taking into account its nature, scale and setting, and whether it could have a significant adverse impact on the purposes for which the area has been designated or defined.
[56] Circular 06/2005 provides further guidance in respect of statutory obligations for biodiversity and geological conservation and their impact within the planning system.
[57] Where areas that are part of the Nature Recovery Network are identified in plans, it may be appropriate to specify the types of development that may be suitable within them.

adequately mitigated, or, as a last resort, compensated for, then planning permission should be refused;

b) development on land within or outside a Site of Special Scientific Interest, and which is likely to have an adverse effect on it (either individually or in combination with other developments), should not normally be permitted. The only exception is where the benefits of the development in the location proposed clearly outweigh both its likely impact on the features of the site that make it of special scientific interest, and any broader impacts on the national network of Sites of Special Scientific Interest;

c) development resulting in the loss or deterioration of irreplaceable habitats (such as ancient woodland and ancient or veteran trees) should be refused, unless there are wholly exceptional reasons[58] and a suitable compensation strategy exists; and

d) development whose primary objective is to conserve or enhance biodiversity should be supported; while opportunities to incorporate biodiversity improvements in and around developments should be encouraged, especially where this can secure measurable net gains for biodiversity.

176. The following should be given the same protection as habitats sites:

a) potential Special Protection Areas and possible Special Areas of Conservation;

b) listed or proposed Ramsar sites[59]; and

c) sites identified, or required, as compensatory measures for adverse effects on habitats sites, potential Special Protection Areas, possible Special Areas of Conservation, and listed or proposed Ramsar sites.

177. The presumption in favour of sustainable development does not apply where the plan or project is likely to have a significant effect on a habitats site (either alone or in combination with other plans or projects), unless an appropriate assessment has concluded that the plan or project will not adversely affect the integrity of the habitats site.

Ground conditions and pollution

178. Planning policies and decisions should ensure that:

a) a site is suitable for its proposed use taking account of ground conditions and any risks arising from land instability and contamination. This includes risks arising from natural hazards or former activities such as mining, and any

[58] For example, infrastructure projects (including nationally significant infrastructure projects, orders under the Transport and Works Act and hybrid bills), where the public benefit would clearly outweigh the loss or deterioration of habitat.

[59] Potential Special Protection Areas, possible Special Areas of Conservation and proposed Ramsar sites are sites on which Government has initiated public consultation on the scientific case for designation as a Special Protection Area, candidate Special Area of Conservation or Ramsar site.

THE PLANNING ACT 2008

THE INFRASTRUCTURE PLANNING (EXAMINATION PROCEDURE) RULES

2010

East Anglia ONE North Offshore Wind Farm

Appendix A12 to the Natural England's Deadline 4 Submission

NE advice on Red-Throated Divers in the Outer Thames Estuary Special Protected Area related to Deadline 3 submissions

For:

The construction and operation of East Anglia One North Offshore Windfarm, a 800MW windfarm which could consist of up to 67 turbines, generators and associated infrastructure, located 36km from Lowestoft and 42km from Southwold.

Planning Inspectorate Reference: EN010077

13th January 2020

Natural England advice on Red-Throated Divers (RTD) in the Outer Thames Estuary Special Protected Area (OTE SPA) related to Deadline 3 submissions

This document is applicable to both the East Anglia ONE North and East Anglia TWO applications, and therefore is endorsed with the yellow and blue icon used to identify materially identical documentation in accordance with the Examining Authority's (ExA) procedural decisions on document management of 23rd December 2019. Whilst for completeness of the record this document has been submitted to both Examinations, if it is read for one project submission there is no need to read it again for the other project.

1) Introduction

This document provides Natural England's statutory advice based on points raised in the following documents submitted by the Applicant at Deadline 3:

- REP3-40/41 Offshore In-principle Monitoring Plan
- REP3-49 Displacement of Red Throated Divers (RTD) in the Outer Thames SPA
- REP3-052 Project Update Note
- REP3-070 Applicants' Comments on Natural England's Deadline 2 Submissions
- REP3-073 Offshore commitments
- REP3-074 Best Practice Protocol for minimising disturbance to RTD
- REP3-084 Written Summary of Oral Case (ISH1)

Summary

i) Natural England welcomes the undertaking of the additional analysis of red-throated diver displacement from the Outer Thames Estuary SPA. But as set out in Natural England's REP3-117 it is unfortunate given Natural England's fundamental concerns, that this assessment wasn't included as part of the original application and/or prior to examination commencing. It is Natural England's view that these issues are better dealt without outside of the constraints of examination.

ii) Natural England raises fundamental issues regarding the Applicant's modelling approach. These relate to inclusion of aerial surveys without corrections for

observer bias, application of shipping lane data and pseudo-replication for spatial and temporal parameters which is likely to **result in displacement up to 7km being an underestimate.**

iii) As the Applicant is basing their conclusions on their modelling approach, which we consider requires further consideration and validation, we believe the conclusions in Tables 5, 7 and 10 of REP3-049 to be unreliable. **Until the modelling approach has been validated, and the issues around treatment of the visual aerial surveys have been addressed, Natural England cannot agree with any of the conclusions.**

iv) The modelled predictions of displacement are completely inconsistent with all empirical studies of red-throated diver displacement recorded at windfarms within or near the Outer Thames Estuary SPA. **Therefore, until such time that these predictions are validated, we consider it wholly inappropriate to assume that the reduction of relative density within the windfarm is 33%**. As stated in MacArthur Green (2019) evidence suggests that displacement of red-throated divers tends to be around 80-100% from within offshore wind farms. **Therefore, until there has been validation of model predictions, the 'effective area of displacement' within windfarms themselves should be considered as the upper end of the range for within windfarm displacement for assessment purposes i.e. 80-100%**.

v) Whilst East Anglia Two will have less of an impact on red-throated diver in the Outer Thames Estuary SPA than East Anglia One North; we disagree that there will be no displacement from East Anglia Two. **We advise that the questions raised around the modelling approach are addressed before effects from East Anglia Two can be ruled out.**

vi) There is clear evidence that the distributions of the red throated diver within the site have already changed as a result of windfarms that are already constructed within the Outer Thames Estuary SPA, and it is likely that if East Anglia One North is built just 2km away from the SPA boundary, there will be additional effects on the distribution of red-throated diver. The key issue here is the effect of the presence of additional windfarms inside or close to the boundary of the SPA has on the distribution of divers and how this may impact on the ability of the SPA to support the same number and distribution of birds as it otherwise would. **Given the expected displacement of red-throated divers in the Outer Thames Estuary SPA due to existing (and planned) offshore wind development, further displacement should be avoided.**

vii) If the proposed windfarm was to be moved 10km away from the Outer Thames Estuary SPA boundary this is likely to negate any significant effects of displacement.

viii) The approach taken by the Applicant to quantifying "effective habitat loss" ignores the issue that over the whole of the area of overlap defined by whatever buffer is considered appropriate (>10km in our opinion), the density of divers and so the ability of those sea areas to support the qualifying feature **will be reduced** to some degree. Therefore, not-withstanding the disagreement on the modelling outputs Natural England advises that displacement 6-7% of the SPA population is a significant effect.

ix) Natural England has set out its legal submission in Deadline 4 Appendix A15 our views on fundamental flaws in the legal position set out by the Applicant in Section 4 and 5 of the Displacement of RTD in the OTE SPA document [REP-049] which should be read along-side the technical advice provided in this response.

Detailed Comments on Displacement of Red-Throated Divers in the Outer Thames Estuary Special Protected Area [REP3-049]

1) Buffer - Major Concern

1. Natural England project advice: The Applicant states their view of Natural England's current position, that a robust HRA which assumes that displacement extends up to 10km, is more conservative than our position pre-application. However, this is not the case. We raised the issue that the recent evidence suggests that red-throated diver displacement extends to beyond 10km before application submission as part of the Expert Topic Group meetings in June 2019.

2. In addition, whilst we acknowledge that there is research from the German Bight that indicates red-throated diver displacement is at least to 10km; this was not the only research which informed our position. The London Array final year post construction monitoring report (APEM, 2020), which reported displacement out to 11.5km within the Outer Thames Estuary SPA is of direct relevance and informed our position more than the similar evidence emerging from the German Bight.

3. It is incorrect to state that Natural England's original recommendation was to employ a buffer of 4km. For EIA purposes we recommend that an assumption that there is 100% displacement out to 4km is used. However, since pre-application we have consistently advised that the HRA for the Outer Thames Estuary SPA needs to take account of the evidence that displacement of red-throated divers is in excess of 10km.

4. Adoption of 2km buffer: Natural England welcomes the Applicant's commitment to the provision of a 2km buffer between East Anglia One North and the SPA boundary. However, we note and agree with the Applicant's position that the 2km buffer reduces the impact, but the proposed buffer does not fully mitigate the impact of displacement to an acceptable level.

5. Locating OWFs away from RTD SPAs: We also acknowledge that there will not be complete avoidance within the buffer, instead there is a gradual decline in displacement with increased distance from the windfarm. However, whether the displacement is 7km, as proposed by the Applicant's modelling, or 11.5km as predicted by the London Array monitoring, **the area affected is significant**. The advice in: *"Review of ornithology constraints for Offshore Wind Leasing in Areas 3 (Yorkshire Coast) and 4 (The Wash)"* Report to The Crown Estate (March 2019) by MacArthur Green states:

> *"Since offshore wind farms can displace red-throated divers up to distances that in some extreme cases appear to exceed 10 km from the turbines, it may be prudent to trim the inshore boundary of Regions 3 and 4 so that these are a minimum of 10 km from the outer edge of Greater Wash SPA."*

6. Natural England agrees with this advice, and although this is in the context of the Greater Wash SPA, we advise that the same approach needs taken here to avoid an Adverse Effect on Integrity (AEoI) of the Outer Thames Estuary SPA.

2) Visual Aerial Surveys - major concern

7. A key issue is that the 2002-2008 visual aerial data set appears to be treated the same as 2013 and 2018 survey data that was collected using digital photographic aerial methods. The visual aerial survey represents an accurate distribution of red-throated diver at the time when all (apart from one) offshore wind farms had not

been constructed. However, it cannot be considered to be an accurate reflection of the absolute numbers of red-throated diver in the SPA at the time, due to visual aerial methods under-estimating true numbers. APEM (2010) carried out comparisons between visual and digital survey methods and reported that digital stills reveal up to 6.5 times as many birds as the visual spotter method.

8. Therefore, the Applicant has not accounted for the likelihood that, in comparison with post construction surveys (conducted by digital aerial photography), the abundance figures in and around the windfarm footprints during the pre-construction period are artificially low due to the use of visual aerial survey methods at that time, there is a risk that the predicted level of displacement will be significantly underestimated.

9. Whilst we recognise that the APEM (2020) monitoring is of course a smaller area, and uses a number of environmental co-variates, the key issue is the baseline surveys in 2009 were undertaken using digital aerial surveys, and therefore comparable to post construction surveys which used the same survey platform. In contrast, the Applicant's approach combines data from different survey platforms, but without considering any correction factors to correct for the perception bias in the earlier visual surveys.

10. Recognising the change in survey platform as an issue Natural England recommended that an approach that considered change in relative densities inside the windfarm footprint and outside needed to be considered.

3) Pseudo-replication – major concern

11. It is unclear how the strengths of effects are being defined. This is particularly in lieu of the fact that there was uncertainty around the partial plot of distance to windfarm in regards to the confidence limits around the presented relationship. As it stands, the confidence limits around the partial relationship between abundance and distance to wind farm includes a straight line through 0 on the y-axis, which suggests that the actual relationship as presented may not exist, or may not be strong.

12. We note that the relative diver distributions in Appendix 1 figure 6a and 8a are very similar. The remarkably similar distributions between factuals and counterfactuals

(Appendix 1 figures 7 and 8) are likely due to the fact that the covariates used are mostly static, and if distance to windfarm is not strongly influencing predictions, removing it or changing it as a parameter should not greatly impact predictive output. Counterfactuals were created here to act as a baseline for which to compare model predictions. This was done by simply changing the distance to windfarm parameter to represent the environment between 2002 and 2006. However, a year covariate and a spatial smoothing covariate were also included in the model. The concern is that there is a signal in the spatial and year parameters that confounds the distance to windfarm parameter because distance to windfarm is a covariate that is made up by the year and spatial elements, thus potentially incorporating an element of pseudo-replication in the model. Therefore, the impact of the windfarms, which is captured by inclusion of the year and spatial parameters, is being passed to the counterfactual. It is therefore unlikely that they actually represent a true baseline to compare against. It is recommended that this potential source of pseudo-replication be investigated and removed if possible.

4) **Displacement effect** – major concern

13. The predicted abundance of RTD displayed in Tables 1 and 2 of REP3-049 is surprising. The comparison of the empirical data in the Applicant's review (in Table 1.1 Appendix 2) stated the level of displacement within the actual windfarm footprint ranged typically between 78% and 95% displacement (Table A below includes all the studies from within or near the Outer Thames Estuary SPA).

Table A: Results of studies near or within the Outer Thames Estuary SPA that had reported effects on red-throated diver within the windfarm footprint

Study	**Location**	**Reported effects on red-throated diver within the offshore wind farm**
Percival (2013): Thanet Offshore Wind Farm Ornithological Report 2012-13	Near Outer Thames Estuary SPA	Compared to the preconstruction phase, within the offshore wind farm an 82% decline in red-throated diver abundance was recorded during construction, and 73% during operation.
Gill et al. (2018): Greater Gabbard offshore wind farm	Near Outer Thames Estuary SPA	Compared to the 0-4km buffer around the offshore wind farm, red-throated

Elston et al.(2016): Greater Gabbard Offshore Wind		diver densities within the offshore wind farm declined 83% between preconstruction and construction.
Percival & Ford (2017): Kentish Flats Offshore Wind Farm Extension: Ornithological survey annual report, October 2016 – March 2017.	Within Outer Thames Estuary SPA	The mean encounter rate within the offshore wind farm dropped from 0.55 birds/km prior to construction, to 0.03 in the first operational year and 0.13 in the second operational year. This was equivalent to reductions of 95% and 76% on the preconstruction baseline.
NIRAS Consulting (2016): Gunfleet Sands 1&2 Offshore Wind Farms	Within Outer Thames Estuary SPA	Survey results indicated a 90% reduction in recorded abundance of divers within the offshore wind farm between preconstruction and construction/operation. Statistical analysis indicated the difference between years was significant.
Percival (2014): Kentish Flats Offshore Wind Farm: Diver Surveys 2011-12 and 2012-13	Within Outer Thames Estuary SPA	Displacement of red-throated divers from within the offshore wind farm was apparent in all seven years of operational monitoring. The magnitude of this effect was calculated to be between 89% and 94% depending on the comparisons undertaken.
McGovern et al. (2020) Final Ornithological Monitoring Report for London Array Offshore Wind Farm	Within Outer Thames Estuary SPA	The proportion of divers displaced from the London Array footprint was estimated to be approximately 78% and 55% for during and post-construction respectively.

14. **Therefore, we advise that Tables 1 and 2 of REP3-049 which indicate that the within windfarm reduction is only ~33% is not consistent with the empirical studies carried out in or near the Outer Thames Estuary SPA (Table A above).**

15. The Applicant's modelling has not found a similar diver response (reported elsewhere) when considering within the windfarm area. Other studies in the Outer Thames Estuary, the Greater Wash and the German Bight have all reported much greater levels of displacement within the windfarms themselves. Whilst we agree that results in one region are not automatically transferable to another region; this is not the situation here. **Empirical data from windfarms impacting upon the same SPA namely the Outer Thames Estuary SPA are demonstrating significant differences in diver responses when compared to this (theoretical) modelling**.

16. Contrary to the Applicant's opinion, the observations do not appear to be similar to the London Array post construction monitoring (APEM, 2020). Firstly, the London Array data; although lower than many of the studies in Table A, a 55% reduction within the windfarm is still considerably higher than the 33% found by the Applicant's modelling approach. This disparity, together with the fact that no cross-validation efforts were made to determine the predictive performance of the model, leads Natural England to question whether **the model is underestimating the displacement effects**

5) Validation of Model predictions – major concern

17. The issue of the un-validated model prediction, that the percentage reduction within the windfarm is approximately 33%, does not relate to East Anglia One North because the proposed array is located outside of the SPA. However, it does mean the total displacement that the SPA is already subjected to is massively under-estimated.

18. As stated above, there are questions over the modelling approach. In particular, the modelling approach does not include any attempt to validate the model predictions, for example by: i) comparison of modelled densities of divers in certain places of particular interest e.g. in and around windfarms with the actual densities recorded or, ii) formal cross validation by exclusion of some of the survey data from the dataset used to construct the models and comparison of the densities

generated by those models with the withheld density data. We consider that model validation tests of this kind are necessary to establish the robustness of the model's predictions and so the degree of confidence can be placed in the findings. Once confidence has been established in the modelling approach, then an assessment of the extent of area affected and numbers of birds predicted to be displaced can be undertaken.

19. In addition, we note that the modelling undertaken in the German Bight (Vilela 2020) uses a more sophisticated Bayesian approach, which may also be appropriate for these projects and help address concerns

20. Given the questions around the validity of the modelling approach we suggest that a range of displacement figures are presented, based on:

- varying spatial extents of effect (including 7km from the Applicant's modelling, but also up to 12km, to reflect the evidence from the London Array monitoring).

- varying magnitudes of displacement and associated gradients with increasing distance (including the Applicant's modelled displacement of 33% within the windfarm footprint, associated gradient out to 7km and up to 100% within the windfarm area and associated gradient out to 11.5km to reflect the empirical studies that have reported much higher levels, typically 80-100% within windfarm footprints).

6) Conservation Objective to maintain diver abundance – major concern

21. As stated in REP1-172, the mortality rate as a result of displacement is not the main the area of concern to Natural England in relation to the Habitats Regulations Assessment. Even if no birds died as result of displacement, the fact remains that the distribution of the divers within the SPA has been changed and will most likely be further changed by the current developments, probably (given the absence of evidence of habituation) on a continuing/lasting basis, and the area of SPA that can support divers will have been reduced.

22. In addition, it is impossible to make direct comparisons between the abundance estimates due to the different survey platforms. When the Outer Thames Estuary SPA was classified in 2010, the best population estimate of the overwintering red-

throated diver population was 6,446 individuals (1989 to 2006/07, peak mean estimate). Technology and survey techniques have vastly improved since, and digital aerial imagery has become the new survey standard (HiDef Aerial Surveying Limited 2018). This has allowed more accurate counts of red-throated diver, and suggests that previous counts may have been underestimates (Goodship et al 2015). There is no way of knowing what the abundance of red throated divers in the SPA would have been prior to the construction of any of the windfarms if those surveys had been carried out using digital photographic means. Therefore, there is insufficient robust data to state confidently what impact the OWF's have had on RTD displacement and mortality and any impacts are inconsequential due to the 'apparent' changes in population. Natural England's advice is to consider the changes in relative abundance inside and outside of the windfarm areas. This will require an alternative approach to treating the visual and digital aerial data in the same way. Furthermore, by focussing on the mortality arising from displacement and the impact of that on population abundance, the assessment misses the key issue which is the need to reduce the frequency, duration and / or intensity of disturbance

7) Conservation Objective to maintain diver distribution – major concern

23. We note the Applicant's point that the number of divers that would be displaced from within and around East Anglia One North/East Anglia Two may be relatively small compared to the total number of divers recorded across the entire SPA. However, maintaining the abundance of divers is not the only Conservation Objective; there is also the objectives to maintain the distribution of divers, and to maintain the area of supporting habitat. The percentage of the SPA affected is significant. For East Anglia One North/East Anglia Two alone out to 8km is 9,019.56 hectares (2.3%), 10km is 12,867.84 hectares (3.28%). In combination with other plans and projects already built, the area of the SPA affected by some degree of displacement at, 8km is 158,033 hectares (40.2%) and 10km is 195,691 hectares (49.86%). We acknowledge that there is not complete displacement, but a gradual decline in the displacement levels the further divers get from a windfarm. These calculations do not account for the 2km buffer at EA1N, but there is still a significant area of the SPA subject to a change in the distribution of divers and a further reduction in available habitat when accounting for the buffer.

24. The Applicant suggests that a similar amount of divers are predicted to be displaced compared to their modelling approach when the alternative approach of 100% displacement of birds out to 4km is considered. With regard to this, firstly there are questions over whether the Applicant's modelling approach underestimates the levels of displacement due to not accounting for the change in survey platform. Therefore, the level of displacement is likely to be underestimated due to the issue of not correcting for the different survey platforms. Secondly, the SPA Conservation Objectives are not simply about maintaining abundance targets. There are also objectives to maintain the distribution of the interest feature throughout the SPA, and to maintain the extent of supporting habitat. Finally, the post construction monitoring from London Array reports that the number of divers displaced in a gradient out to 11.5km was greater than the assumption that 100% of birds were displaced up to 4km.

25. As previously stated, even if no birds die as a result of the displacement, a separate but equally important issue, dictated by the full suite of conservation objectives for the SPA is the continuing (and apparently permanent) change in distribution of the interest feature. **This compromises the ability of the supporting habitat within the SPA to continue to support the same numbers and distribution of birds as before**.

8) Ecological consequences of displacement – major concern

26. We acknowledge that the likely consequences (lethal or otherwise) of displacement that results from the concentration of more birds into a smaller area of sea distant from all windfarms is not known and may indeed be small. However, while many of the statements made in this section of the report in support of the conclusion that the effects will be minimal and of no significance are justified, some are not and so the case is overstated. For example, it is incorrect to state that "*in the absence of highly aggregated regions for this species, it appears unlikely that existing or planned windfarms occupy sites of particular importance for this species….. hence the first mechanism above (exclusion from preferred foraging areas) is not considered to be applicable*".

27. By definition the SPAs include the areas of highest average density, usually assessed over a period of several years, and within those there are often certain areas that persistently hold the highest densities. One might infer from this that

the SPA boundary includes the most suitable habitat for the species and that within those there are areas of particularly high suitability. Certain windfarms have already been constructed within the most heavily used areas and have excluded birds from those preferred areas.

28. The Applicant has stated that red throated divers appear to be capable of a high degree of mobility in winter, and therefore they will be able to find alternative foraging areas, albeit in some cases distant from the original area of displacement. This may be true, but this fails to acknowledge the critical point that in the context of this Habitats Regulations Assessment, displacement of birds from inside to outside an SPA that would otherwise support them means that the integrity of the site has been compromised. The site would no longer be making the contribution that it otherwise would to the favourable conservation status of the species or the coherence of the Natura 2000 network for that species.

29. Furthermore, in the context of this Habitats Regulations Assessment, all the Conservation Objectives of the SPA must be considered. In this case the Conservation Objectives include an objective to maintain or restore the distribution of qualifying features within the site. Furthermore, the Supplementary Advice on Conservation Objectives for the red-throated diver feature of this SPA specify a target to: *"**Reduce** the frequency, duration and / or intensity of disturbance affecting roosting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed"*. It may be that no birds at all die as a result of the displacement, but it is in the light of these Conservation Objectives it is still possible that an AEoI on the SPA will result from one or more of the other conservation objectives not being fulfilled.

9) Legal protections afforded to the Outer Thames Estuary SPA – major concern

30. Baseline: The reference to a 'baseline' for assessment of the effects of a proposed plan or project is not found in the Habitats Directive or either of the two domestic statutory instruments and leads to incorrect conclusions. We note that the inclusion of the wind farms within the OTE SPA that received consent prior to the existence of the SPA as part of the in-combination assessment of EA1N and EA2 is included. However, Natural England advise that these existing windfarms (and those that became operational before the current RTD population figures were

established) should be included as a matter of law and not 'for illustrative purposes' only.

31. Significance of impact: The use of the term 'significance of disturbance' (paragraph 69), and the suggestion that it should be considered by reference to the 'objectives for the whole region or an EU Member State' is incorrect.

32. We agree that it is necessary to consider the significance of the disturbance resulting from the projects, but in the context of the effect of disturbance on the integrity of the SPA itself, and not in a regional or other spatial context. Natural England's view is that there is strong evidence to confirm that the presence of wind turbines does constitute significant disturbance to red-throated divers. **This appears to be long term based on the lack of any evidence showing diver habituation to the presence of offshore windfarms.**

33. As per our Deadline 1 Appendix A4 [REP1-172] submission we advise that the Examining Authority base their assessment on an in-combination assessment that includes all projects that were not constructed at the time of the surveys to inform the SPA notification i.e. those projects constructed after 2002-2008. It was the spatial variation in the birds' density at that time which informed the analysis that determined the size and shape of the SPA.

34. Natural England's Supplementary Advice on Conservation Objectives for the Outer Thames Estuary SPA can be found at: https://designatedsites.naturalengland.org.uk/Marine/SupAdvice.aspx?SiteCode=UK9020309&SiteName=outer+thames+estuary&SiteNameDisplay=Outer+Thames+Estuary+SPA&countyCode=&responsiblePerson=&SeaArea=&IFCAArea=&NumMarineSeasonality=3

35. Within the 'disturbance caused by human activity' attribute for red-throated diver, the supporting notes state:

> *"Disturbance should be judged as significant if an action (alone or in combination with other effects) impacts on (water)birds in such a way as to be likely to cause impacts on populations of a species through either:*
>
> - *changed local distribution on a continuing basis; and/or*
> - *changed local abundance on a sustained basis; and/or*

- *the reduction of ability of any significant group of birds to survive, breed, or rear their young."*

36. It is due to **a changed local distribution on a continuing basis** that we consider there is significant disturbance. The key point is that the phrase "likely to cause impacts on populations of a species…" does NOT say "population abundance or size". This means that significant disturbance can be disturbance that impacts the population through changing its local distribution on a continuing basis – regardless of what that means for the population's size and abundance

37. Natural England acknowledges that the abundance objective is likely to be maintained. However, our concerns are whether the distribution of the divers are changed on a continuing basis, and whether the SPA is able to support divers throughout the whole of the SPA. The key point is that the disturbance attribute has a target to "restore" and so that is not favourable and that relates to the headline objective to maintain or restore the distribution of qualifying features within the site – which is a separate high level Conservation Objective to the population one.

10) In-combination – major concern

38. As stated in our advice submitted at Deadline 1 (REP1 -172) the data regarding RTD that effectively determined the current size, shape and boundary of the SPA was derived from surveys undertaken before all but one of the relevant offshore windfarms were constructed (the earliest survey data even preceded the operation of Kentish Flats). Therefore, we advise that an in-combination assessment of the level of displacement should include all the projects that have become operational since that baseline understanding of the distribution of RTD was established, namely:

• London Array
• Gunfleet Sands I, II and III
• Kentish Flats and Kentish Flats Extension
• Greater Gabbard
• Thanet.

39. With regards to East Anglia Two, the monitoring results from London Array post construction monitoring suggest that displacement distances are up to 11.5km. Until the Applicant's modelling addresses our concerns raised in the above sections, we continue to advise that East Anglia Two and the above projects are included in the in-combination assessment.

11) Shipping data – moderate concern

40. For shipping we note that there was a choice to convert shipping traffic into a rasterized map. This decision needs more consideration as there does not seem to be much precedent for this in the literature. The concern with the use of shipping data like this is that there are areas where shipping traffic is simply 0, and those pixels could fall directly next to a shipping lane, which means that a gradient effect could be missed. Convention is to use distance to shipping lanes.

41. In addition to concerns detailed above regarding treating all shipping as a static variable and not as distance to shipping lane; we are unclear whether it is appropriate to assume the same level of shipping in 2012 compared to 2002 – 2008.

12) Assumed Increases in Red Throated Diver Numbers – moderate concern

42. We acknowledge that estimated abundance of the RTD population within the SPA has increased significantly over the period within which several windfarms have been constructed within it. However, it is important not to infer from this that it is a fact that the actual population abundance has significantly increased. This is because of the fundamental shift in the survey method used over that period. That the two digital surveys in 2018 yielded an average population abundance greater than the average in 2013 also cannot be used to infer with any confidence that the population has increased over that time period. The first of the two surveys conducted in 2018 yielded a population abundance estimate that was lower than on either of the two surveys in 2013. Only the last survey in 2018 yielded a number in excess of 20,000. The fact that this was recorded was a matter of chance, and similar short-lived peaks in abundance are highly likely to have been missed in previous years as surveys are only a snap-shot in time. Therefore, we advise that the Applicant appears to be confusing abundance and distribution targets.

13) Model evaluation – minor concern

43. We note that nine models were evaluated, using three different error structures (Poisson, Tweedie, negative binomial). This is convention with General Algebraic Modelling System (GAMs) and the authors have treated this appropriately. However, one point here is that the Poisson distribution as used in the code (i.e., the Poisson() family) may not necessarily be the most appropriate for zero-inflated data in all cases. Best practice for zero-inflated Poisson data would be to use the ziP() family (i.e., zero-inflated Poisson). Whilst it is unlikely that this would drastically change the outputs of the modelling, it should be explored in case it does help to explain a little more of the variance in the data.

14) Effects of Bathymetry – minor concern

44. It seems that bathymetry has a strong influence on predictions across the whole site (i.e. the predicted output closely matches the patterns in bathymetry). Thus, interpretation of these partial dependence plots needs to be done with care because they are not a representation of the exact relationship in space, just a partial relationship when taking into account the other parameters. Therefore, in areas where distance to coast is <15km, if another parameter has a stronger positive effect than the negative effect of distance to coast in that range, the model could still predict higher densities.

15) Comments on Appendices 1 - 3 of REP3-049

45. Advice provided for the main body of the document also apply to Appendices 1 and 3

46. Appendix 2: The literature review appears to be a thorough review of the available evidence relating to red-throated diver displacement, although it is noted that the Final Year Ornithological Report for London Array is within the References on page 32, but not included in the Literature Review. It should also be noted that the date of the up to date version of that report is 2020, as the 2018 version in the References on page 32 does not include Natural England and MMO's comments.

Interim comments on Offshore In-principle Monitoring Plan [REP03-040]

47. Natural England agrees that ornithological monitoring should be targeted to address residual impacts, evidence gaps or uncertainty of most relevance to the proposed East Anglia ONE North project and the specific species. We agree that should focus on monitoring the extent of displacement on red-throated diver and undertaken as part of a pre- and post construction monitoring programme. This will be particularly important if a design is consented where the buffer is less than 10km or less than the to be agreed modelled extent of displacement. Natural England will provide at Deadline 5 further advice on specific hypothesis/positions to be tested/validated by the post construction monitoring

48. Natural England's advice is that EA2 should be included as part of an in-combination assessment for the Outer Thames Estuary SPA. Therefore, comments made by Natural England in relation to the East Anglia ONE North IPMP, also apply to EA2.

Comments on Offshore commitments [REP3-073]

49. At the Expert Topic Group meeting on 20th June 2019, the Applicant provided details of proposals to move the boundary of EA2 (which previously was immediately adjacent to the SPA) to 8.3km away from the SPA at its nearest point. The Applicant's explained that while the site reduction was primarily to mitigate seascape impacts, it has also resulted in the EA2 windfarm site now being further from the Outer Thames Estuary (OTE) Special Protection Area (SPA) and therefore significantly reducing any displacement impacts from EA2. Natural England acknowledged that there would be now significantly reduced effects on OTE SPA from operational windfarm array displacement from EA2, and advised a similar approach to be taken with EA1N.

50. Whilst, it was possible to reduce the array footprint for EA2 resulting in the nearest turbine being 8.3km from the Outer Thames Estuary SPA; Natural England acknowledges the constraints on the Applicant's options for relocation of EA1N.

51. Natural England notes that the Applicant's view is that the 2km does not mitigate the displacement effects, only reduces them by 8%. Although there is uncertainty

on the exact number of red throated diver that may be displaced, it is clear that the extent of supporting habitat extends well beyond 2km. So, whether the extent of displacement is 7km as the Applicant's suggest from their recent modelling or 11.5km that was concluded from the London Array's monitoring within the Outer Thames Estuary, the buffer needs to be significantly increased from 2km. Without an increase in the distance between the proposed array and the SPA boundary an adverse effect on integrity from the EA1N project alone cannot be ruled out as a result of displacement reducing the ability of the supporting habitat and a change in the distribution of red throated divers on an ongoing basis.

Detailed comments on Best Practise Protocol for minimising disturbance to RTD [REP3-074]

52. Natural England welcomes the submission of the broad principles for a best practice protocol for minimising disturbance to red throated diver. We note that the intention is that the protocol will be adopted and provided as part of the project environmental management plan (PEMP) and to be approved by MMO. Although we note that detail will be included at a later stage, it would be beneficial to provide some additional detail in this document of the expectations and detail that will be required to sign of this mitigation measure prior to commencement of construction for example: -

- How will it be demonstrated that planned works during construction and operation phases are avoiding the sensitive periods between November and March?
- Were it is not possible to avoid works during the sensitive period how will vessel movements be managed to minimise disturbance to SPA features?
- Provided details of particular works when vessels will be required to leave existing navigational routes through the SPA.

53. Low flying helicopter flights over the SPA are also likely to cause disturbance. If the use of helicopters is likely then we advise that is also covered under a protocol for minimising disturbance.

Bird Study (2006) **53,** 1–11

Influence of environmental factors and energetic value of food on Little Tern *Sterna albifrons* chick growth and food delivery

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Capsule Chick diet and early growth parameters differed between birds breeding on natural (sandy beaches) and alternative (salinas) habitats.

Aims To describe diet and growth of Little Tern chicks, to examine the influence of tide and wind speed on food delivered to chicks in southern Portugal, and to determine whether diet and early chick growth parameters differed between birds breeding in natural (sandy beaches) and man-made alternative (salinas) habitats.

Methods Nests were marked and visited regularly. Chicks were ringed and weighed at hatching. Between 12 and 14 nests were enclosed by a fence in order to measure and weigh chicks every day. Prey items delivered to fenced chicks were observed daily from hides, in different tidal periods and wind speed conditions. Prey items dropped in the colonies were collected. Foraging efficiency of adults was evaluated in relation to tide and wind speed. Fresh prey items were collected to determine their energetic content using a calorimeter.

Results *A* chicks grew faster than *B* chicks, but both attained the same weight, tarsus-length and wing-length at 19 days old; *C* chicks attained the same values at day 21. There were significant differences in early growth parameters. *A* and *B* chicks grew faster on sandy beaches than on salinas. The main prey items were *Atherina* spp., *Fundulus* sp. and shrimps in salinas and *Sardina pilchardus, Atherina* spp. and *Belone belone* on sandy beaches. There was no correlation between ingestion rate and chick age, but mean length of prey ingested by chicks increased significantly with age.

Conclusion Tide and wind speed influenced both size and type of prey delivered to chicks. Parents met the increasing energy demands of the growing chicks by adjusting prey size rather than prey delivery frequency. Differences in chick growth between salinas and sandy beaches could be related to both parental quality and diet. The presence of a high energy-content euryhaline fish species in the salinas, together with more sheltered conditions and no tidal influences, may be important in explaining the use of salinas by Little Terns.

Variation in food supply has a marked impact on breeding parameters and productivity of small seabirds such as terns (Monaghan *et al.* 1989), because they possess little spare capacity in their energy budgets and spend a larger proportion of their time foraging compared to other seabird species (Pearson 1968). Variation in food supply has been shown to influence chick growth and productivity (Monaghan *et al.* 1989), due to variation in the availability of fish, in the vicinity of the tern colonies. Prey species can also vary in energy content (Hislop *et al.* 1991, Anthony *et al.* 2000), raising the hypothesis that energy limitation may occur if the most abundant prey available are low in energy (Dahdul & Horn 2003). Of all the temperate tern species Little Tern *Sterna albifrons* is arguably the most estuarine and the species that forages closest to the breeding colonies (Cramp 1985, Fasola & Bogliani 1990). Their estuarine foraging areas are characterized

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by shallow and clear waters (Allcorn *et al.* 2003). Growth and chick food provisioning of estuarine foraging terns are very sensitive to variations in food availability due to wind speed and tide (Frank 1992), but very little is know about the effects of these two factors on Little Tern chick food provisioning.

Norman (1992) suggested that the type and size of available food is the main factor influencing growth rates in Little Tern. However, very little is known about growth of Little Terns in other areas of Europe, especially in southern Europe. Moreover, Norman (1992) did not distinguish between growth rates of the different chicks within a brood.

In Algarve, southern Portugal, where this study was conducted, Little Terns breed in natural (sandy beaches) and man-made alternative (salinas) habitats (Catry *et al.* 2004). Foraging Little Terns feed in natural (estuarine lagoon system and adjacent sea) and man-made (salinas and artificial channels) habitats. Several breeding parameters such as egg size and clutch size were larger in natural habitats, suggesting that these are preferred to man-made alternative habitats (Catry *et al.* 2004).

Here, we describe the diet and growth of Little Tern chicks in Ria Formosa, Algarve, and determine whether diet and early chick growth parameters differed between birds breeding in natural and man-made alternative habitats. On estuaries and coastal areas, tide and wind speed are considered the main factors affecting the accessibility of prey to terns (Frank 1992, Dunn 1972), and we decided to evaluate their influence on the provisioning of Little Tern chicks in Ria Formosa. We examined whether the calorific content of prey delivered to Little Tern chicks differed between birds breeding in sandy beaches and salinas. This was justified because, for instance, birds breeding on salinas were observed foraging in water-tanks used for salt production, where the composition of prey is likely to differ from that available in the lagoon and sea that surround the colonies in natural habitats.

METHODS

Ria Formosa Natural Park is situated in the south coast of Portugal, Algarve, and consists of a complex tidal system of natural and semi-natural channels (lagoon), marshland and barrier islands covering an area of 18 400 ha along 60 km of coast. In the margins of the marshland area, there are man-made salt-gathering tanks (salinas) and extensive fish farms. Close to the Park's northern borders, there are three large towns, Faro, Olhão and Tavira.

Our observations took place in both natural (sandy beaches of the barrier islands Armona, Faro and Barreta) and alternative (salinas around Santa Luzia, Tavira) breeding habitats (Catry *et al.* 2004). The closest foraging areas were the lagoon and the sea for birds breeding in natural habitats, and the lagoon and salinas for those breeding in alternative habitats.

Chick growth

Nests in the salinas were marked in May and June 2002 and 2003, and visited every two days until hatching, and daily thereafter. Chicks were weighed and ringed on the day of hatching (day 0). Their wing-length (to the nearest 1 mm) and tarsus-length (to the nearest 0.1 mm) were measured on the day of hatching and thereafter whenever chicks were found. Fourteen nests in 2002 and twelve in 2003 from Santa Luzia were selected within an area of approximately 1500 m^2 and fenced with hexagonal wire, 0.3 m high and 1 cm^2 mesh, to keep chicks near the nest-site. Fences were about 1.5 m in diameter. Inside the fences, we placed some rocks for use as perches by the parents, as well as a wooden shelter and some vegetation to provide chicks with protection against the sun and possible avian predators. Chicks were designated as *A*, *B* and *C* according to hatching order (when two chicks hatched on the same day, we considered the *A* chick as the one with higher mass, greater egg volume and greater tarsus-length). Each chick (*A, B* and *C*) of a nest was marked, using non-toxic paint, with a different colour on the head and breast. These chicks (n = 32 in 2003 and n = 25 in 2002) were weighed daily between 08:00 and 11:00 hours, until they were 20 to 22 days old. Daily or on alternate days in 2003, the chicks of unfenced nests (see Fig. 2) were also weighed in order to verify the impact of the fences on chick growth.

We verified that mass data of Little Tern chicks for days 0–4 were fitted to a quadratic regression model.[a]

Two parameters were calculated to characterize growth of the 2003 chicks after the age of four days: linear growth rate (LGR) and asymptotic mass (AM). LGR was defined as the slope of a regression line of chick mass against date (Nisbet *et al.* 1995) during the linear growth period (5–14 days). AM was defined as the mean of all masses measured during the period of near-constant mass (15–22 days) (Nisbet *et al.* 1998). Linear growth rate and asymptotic mass were calcu-

lated for all fenced chicks (weighed daily) with at least four and two data points, respectively.

Food delivery

During regular visits to the colonies on sandy beaches (Armona, Barreta, Praia de Faro) and salinas (Santa Luzia I, Santa Luzia II, Arraial Ferreira Neto, Vale Carangueijo) in 2002 and 2003, 596 dropped prey items (this includes all prey that were not ingested) were collected and identified. Observation of food delivery to fenced chicks occurred between 11 June and 5 July 2003. From the fenced nests, ten were selected according to their location. Two portable hides, placed 2–7 m from these nests, were used to observe and identify prey delivered to chicks, using 10 × 40 binoculars. A total of 72 hours of observation were made, divided into periods of one to ten hours (median = two hours), in different tidal phases (low, high, incoming and receding). We considered that each tidal phase lasted three hours (1.5 hours before and after each tide peak). At the beginning of each observation period we measured wind speed with a field anemometer and later classified it into three categories: 0–2, 2–4, >4 m/s. Water surface conditions were evaluated as: calm, small undulation, high undulation.

For each prey item offered to chicks we registered the number of the nest, chick (*A*, *B* or *C*), time, prey species, prey length and fate. The fate of each prey was classified as ingested or not ingested (the latter category included prey robbed by adults, eaten by parents or dropped). We were familiar with the main prey species in the area (Muzavor *et al.* 1993, Sobral & Gomes 1998), having collected prey dropped in colonies and made observations in 2002. Tests to identify prey and establish their length by the several observers were made during the courtship phase. When it was not possible to identify prey species, only the genus or family name was given, or they were classified as non-identified. Gobiidae and Blenniidae were difficult to separate and were included in the same category (Gobiidae/Blenniidae) with the exception of the Gobiidae *Pomatoschistus* sp., which was easily identified. Prey length was determined in relation to the average Little Tern bill length (mean culmen = 29.85 mm, se = ±0.26, *n* = 26 Little Tern adults captured in 2003, at the barrier island of Armona). Throughout this paper, size is used as a synonym of length. 'Bill-units' were used with a division of 0.25 'bills'. Prey items dropped were picked up after each observation period and used to clarify identification of some prey delivered to chicks.

Foraging behaviour in relation to tide

Fifteen transects were made between 12 June and 12 July 2002, and 33 between 15 May and 19 June 2003, to collect data on foraging adults in relation to tide. Transects were made along the salinas of Santa Luzia and adults actively foraging were assigned to habitat type (salinas or lagoon) during four tidal phases (low, high, incoming, receding) in 2003 and two tide categories (low, high) in 2002.

In 2003 we observed 45 adults diving (for at least one minute each) in the lagoon habitat and registered the number of dives per minute and the number of prey captured per minute in relation to tidal phase.

Calorific content

Fresh prey items found in the colonies were measured (standard length, maximum width), weighed, washed with fresh water and frozen. To determine the calorific content, items were thawed and dried in the oven at 60°C. Once in the oven, prey items were weighed daily until constant mass values were obtained, and then each item was crushed to dust in a mortar. A small quantity of the samples (0.058 to 0.200 g) were converted into discs in a press and used to determine the calorific content of each prey species in a PARR 1425 calorimeter. Three discs of each sample were made and the mean value calculated. The calorific content was given by the calorimeter in calories per gram of dry weight (cal/g dry wt) (Henken *et al.* 1986, Fraschetti *et al.* 1994); this value was converted to joules per gram of dry weight (J/g dry wt) by multiplying by 4.187.

Statistical analysis

Data from sandy beaches (Barreta, Praia de Faro and Armona) were grouped. We evaluated differences both in the linear growth rate and the asymptotic mass between *A* and *B* chicks in the salinas during 2003 using Student's *t*-tests. One-way ANOVA, followed by a post-hoc test (Tukey test), was used to test for differences in the early chick growth parameters (*A* and *B* chicks were grouped) among salinas in 2002, salinas in 2003, and beaches in 2003. ANOVA was also used to test the null hypothesis that the mean calorific content is similar between prey species. We used Pearson correlation coefficients to verify: (1) whether the proportion of each prey item dropped in the salinas was similar to that ingested by the chicks; and (2) whether

the proportion of each dropped prey item was similar between 2002 and 2003 (data were arcsine transformed). To test the null hypothesis that the mean percentage (data arcsine transformed) of the main items in the diet of *A* and *B* chicks was similar, we used Student's *t*-tests. A Mann–Whitney test was used to test for differences between delivery and ingestion rates. The relationships between both ingestion rate and length of prey delivered in relation to chick age were examined with Spearman correlations. Differences in the delivery rate due to tide and wind speed were examined using a two-way ANOVA test. The influence of environmental factors (i.e. tide state and wind speed) in the rate and length of prey delivered was tested using Kruskal–Wallis non-parametric ANOVA. Differences in the number of adults foraging in the lagoon in relation to tide were examined with Mann–Whitney (2002) and Kruskal–Wallis (2003) tests. One-way ANOVA, followed by a post-hoc test (Tukey test), was used in capture success data to test for differences between the four tidal phases. A significance level of $P \leq 0.05$ was set a priori for all statistical tests. We present values as mean ± se. All analyses were conduced using STATISTICA version 6.0 (Statsoft 1996).

RESULTS

Chick growth

The growth curves of *A*, *B* and *C* chicks, in terms of chick mass, wing-length and tarsus-length, are shown in Fig. 1. *A* chicks grew faster than *B* chicks but both attained the same weight, tarsus-length and wing-length at 19 days, whereas *C* chicks reached the same weight and wing-length two days later. There were significant differences in the estimated hatching mass (M_0) and the quadratic growth parameter (b) of *A* and *B* chicks among the salinas in 2002, salinas in 2003 and beaches in 2003 (Table 1). On the other hand, the linear growth parameter (a) did not differ significantly among these three categories (Table 1).

In 2003, both linear growth rate (LGR, mean ± se) and asymptotic mass (AM, mean ± se) did not differ significantly between *A* (LGR: 2.53 ± 0.09 g/day, n = 12; AM: 41.00 ± 1.48 g, n = 10) and *B* (LGR: 2.31 ± 0.17 g/day, n = 10; AM: 39.48 ± 1.23 g, n = 6) chicks (LGR: $t_{20} = 1.16$, $P = 0.26$; AM: $t_{14} = 0.71$, $P = 0.49$). Only two out of eight *C* chicks survived in the fenced nests from five days of age onwards (LGR: 2.28 ± 0.04 g/day).

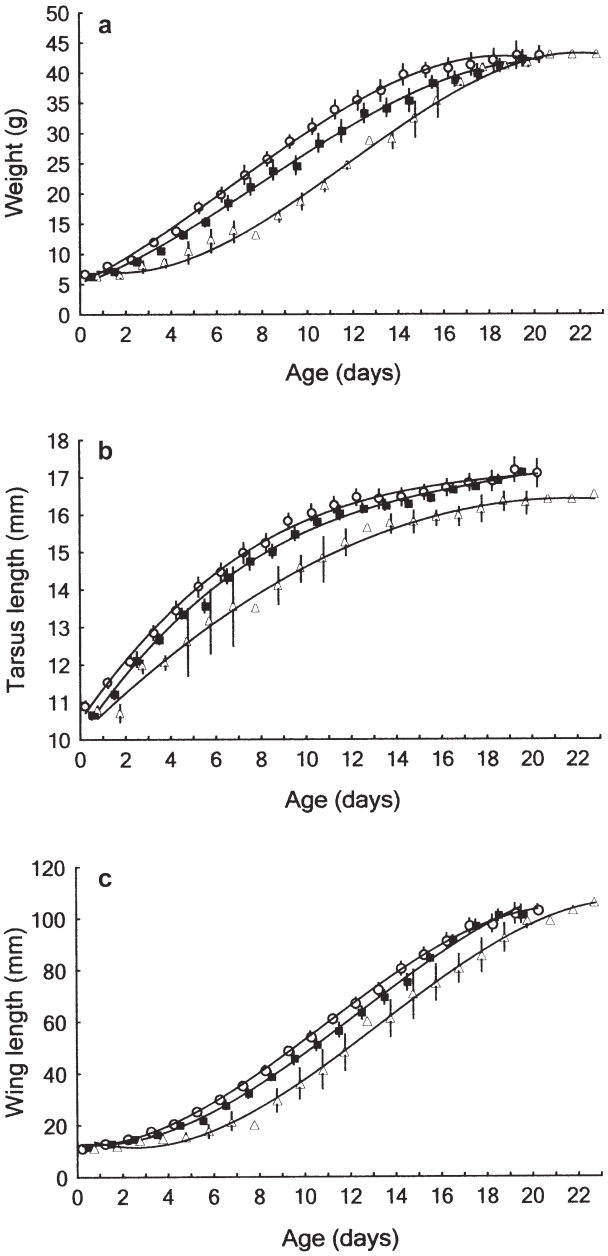


Figure 1. Growth curves (mean ± se; the curves were adjusted by polynomial regression) for Little Tern chicks (n = 32) in terms of: (a) mass; (b) tarsus-length; (c) wing-length. ○, *A* chicks, n = 14; ■, *B* chicks, n = 11; △, *C* chicks, n = 2–7 (median = 2; after day 5 there were only two *C* chicks).

There was no apparent effect of fences on chick growth because the mean mass values of fenced chicks were similar to those of non-fenced chicks (Fig. 2). Although chick growth curves in terms of weight were similar between 2002 and 2003, variability was much higher in 2002 (Fig. 2).

Table 1. Comparison of early chick growth parameters (mean ± se) of Little Tern *A* and *B* chicks for 2002 and 2003 between salinas and beaches (Barreta and Praia de Faro).

	Salinas			ANOVA	
Early growth parameter	2002	2003	Beaches 2003	$F_{2,38}$	*P*
M_0 (g)	6.50 ± 0.14^{ab}	6.54 ± 0.16^{a}	7.17 ± 0.17^{b}	4.16	0.03
a (g/day)	1.07 ± 0.31	0.98 ± 0.15	1.04 ± 0.14	0.06	0.94
b (g/day ^{2})	0.20 ± 0.08^{ab}	0.19 ± 0.04^{a}	0.33 ± 0.04^{b}	3.18	0.05
n	6	17	18		

M_0, Estimated mass on day 0; *a*, linear growth parameter; *b*, quadratic growth parameter. Entries in the same line that do not share the same letter are significantly different (Tukey test, $P \leq 0.05$).

Table 2. Comparison (%) of prey items dropped, ingested and non-ingested by Little Tern chicks in salinas and sandy beaches during June and July.

	Salinas 2002	Salinas 2003			Beaches 2003
Species	Dropped (*n* = 70)	Dropped (*n* = 258)	Ingested (*n* = 857)	Non-ingested (*n* = 45)	Dropped (*n* = 268)
Pelagic fishes					
Atherina spp.	20.0	26.7	19.1	40.0	23.9
Sardina pilchardus	1.4	7.0	7.2	4.4	29.5
Belone belone	4.3	7.4	6.0	11.1	22.4
Scomberesox saurus	0.0	2.3	2.5	4.4	3.0
Mugilidae[1]	8.6	1.9	0.4	2.2	8.6
Shrimps[2]	14.3	20.2	17.5	11.1	0.8
Euryhaline fish					
Fundulus sp.	20.0	19.8	30.2	22.2	8.2
Bottom/demersal fishes					
Soleidae	7.1	3.9	0.1	0.0	0.4
Pomatochistus sp.	1.4	2.3	4.1	0.0	0.4
Gobiidae/Blenniidae	11.4	1.9	4.1	2.2	0.0
Others[3]	11.4	7.0	0.4	0.0	3.0
Unidentified	0.0	0.0	8.5	2.2	0.0

[1]The most common genus of this family in Ria Formosa is *Liza* (Erzini *et al.* 2002). [2]*Paleomonetes* sp. and *Paleomon* sp. [3]Species with $n < 6$ items (*Trachinus vipera, Dicentrarchus* sp., *Loligo* sp., *Raja clavata, Microchirus boscanion, Nerophis ophidion, Engraulis encrasicolus, Symphodus melops, Diplodus* sp.).

Food delivery

The main prey items delivered to chicks and found dropped in the colonies were the fish *Atherina* spp., *Fundulus* sp. and the Shrimps *Paleomonetes* sp. and *Paleomon* sp. in salinas and the fish *Sardina pilchardus*, *Atherina* spp. and *Belone belone* on sandy beaches (Table 2).

There was a significantly positive relationship between the percentage of each prey item delivered to chicks and the percentage of the same prey item found dropped in the salinas in 2003 ($r = 0.68$, $P = 0.02$, $n = 12$). This indicates that prey dropped provided a reasonable estimate of Little Tern chick diet. There was also a positive correlation between the prey found dropped in 2002 and 2003 in the salinas ($r = 0.76$, $P = 0.00$, $n = 12$). However, in terms of general prey types, bottom/demersal fishes were about three times less abundant in 2003 than in 2002 (Table 2), whereas pelagic fishes were significantly more abundant in 2003 (45.3%, $n = 258$) than in 2002 (34.3%, $n = 70$, $\chi^2_1 = 7.44$, $P = 0.006$).

The mean percentage (± se) of *Fundulus* sp. delivered to *A* (16.61 ± 2.96) and *B* (18.07 ± 4.17) chicks did not differ significantly ($t_{16} = 0.29$, $P = 0.77$). The same occurred for *Atherina* spp. delivered to *A* (17.82 ± 1.75) and *B* (19.47 ± 2.96) chicks ($t_{16} = 0.50$, $P = 0.62$), and for shrimps delivered to *A* (12.38 ± 5.90) and *B* (16.28 ± 5.12) chicks ($t_{16} = 0.49$, $P = 0.63$).

Overall mean values (± se) of the different rates for all chicks were: 1.44 ± 0.10 feedings/chick/hour (fd/ch/h) ($n = 112$) for delivery rate by adults, 1.39 ± 0.10 fd/ch/h ($n = 106$) for the ingestion rate and 0.04 ± 0.01 fd/ch/h

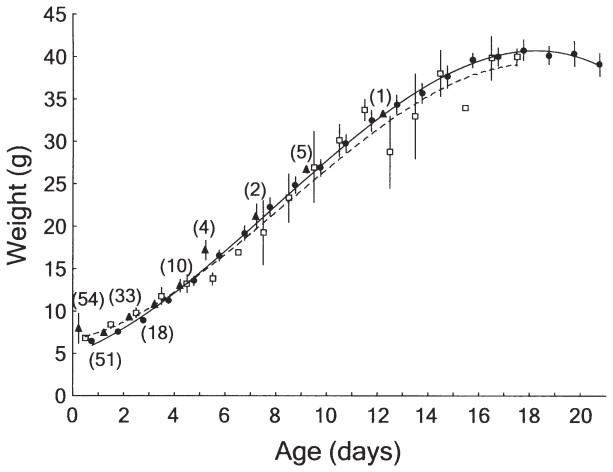


Figure 2. Growth curves (mean ± se; the curves were adjusted by polynomial regression) of fenced *A* and *B* chicks for 2003 (n = 30) (●–●) and 2002 (n = 2–21, median = 4) (□–□) in the salinas, and (▲) unfenced *A* and *B* chicks for 2003. Sample size for unfenced chicks of each age is indicated beside the points.

(n = 6) for the non-ingested prey rate items. The delivery and ingestion rates were very similar (Mann–Whitney: Z = 0.39, P = 0.70, n = 44). The mean rate of food delivered to *A*, *B* and *C* chicks of all ages was 1.38 ± 0.10 fd/h (n = 101), 1.26 ± 0.10 fd/h (n = 86) and 1.16 ± 0.24 fd/h (n = 27), respectively. There was no correlation between ingestion rate and chick age for *A* and *B* chicks (r_s = –0.26, P = 0.25, n = 22, Fig. 3) but mean length of prey ingested by chicks increased significantly with age (r_s = 0.81, P = 0.0001, n = 22, Fig. 3).

Two environmental variables (wind speed and water surface condition) were positively correlated in this study (r_s = 0.78, P = 0.0001, n = 20). Considering all prey items delivered to chicks, there was no significant difference in the delivery rate among the four tide states (Kruskal–Wallis: $H_{3,170}$ = 3.75, P = 0.29) nor with different wind categories ($H_{2,112}$ = 0.72, P = 0.70). A two-way ANOVA on the effect of these two variables on the delivery rate showed no significant influence of tide ($F_{3,170}$ = 0.15, P = 0.93), wind speed ($F_{2,170}$ = 2.72, P = 0.07) and no interaction ($F_{6,170}$ = 0.57, P = 0.75).

The mean length of prey delivered to chicks increased significantly with chick age for *Fundulus* sp. (r_s = 0.75, P = 0.0001, n = 21) and shrimps (r_s = 0.67, P = 0.003, n = 18). Other prey species presented a positive but non-significant relationship: *Atherina* spp. (r_s = 0.42, P = 0.06, n = 20), Gobiidae/Blenniidae (r_s = 0.16, P = 0.56, n = 15), *Scomberesox saurus* (r_s = 0.46, P = 0.14, n = 12), *Belone belone* (r_s = 0.34, P = 0.17, n = 18), *Sardina pilchardus* (r_s = 0.05, P = 0.86, n = 14). *Pomatoschistus* sp. (r_s = –0.32, P = 0.24, n = 15) presented a negative but not significant correlation.

Effects of tidal phase and wind speed on the delivery of specific prey species

For both 2002 and 2003, there was a significant difference in the number of adults foraging in the lagoon habitat among the different tidal phases. In 2003, the number of foraging adults during low tide was significantly higher than during the other three tidal phases. In the same year, the number of dives per minute in the lagoon habitat at high tide was significantly lower than during receding tide but prey capture success did not differ among the four tidal phases (Table 3).

The wind speed apparently affected the delivery rate of *Pomatoschistus* sp. (Mann–Whitney: Z = 2.27, P = 0.02, n = 17), with more prey delivered in calm conditions (0–2 m/s). Tide influenced the delivery rate of *Scomberesox saurus*, with more items delivered during high and receding tides (Kruskal–Wallis: $H_{3,17}$ = 9.19, P = 0.03), and *Sardina pilchardus*, with more items delivered during rising tide (Kruskal–Wallis: $H_{3,33}$ = 7.65, P = 0.05, Fig. 4).

Table 3. Comparison of the mean number of foraging adults and prey capture success (mean ± se, with sample size in parentheses) in the lagoon habitat among the four tide states.

	Tide					
Foraging behaviour	Low	High	Incoming	Receding	Statistic	*P*
Adults/transect						
2002	10.9 ± 1.64 (10)^a	3.57 ± 0.81 (7)^b			Z = 2.68	0.01
2003	5.50 ± 0.76 (6)^a	0.67 ± 0.49 (6)^b	1.89 ± 0.70 (9)^b	3.67 ± 0.86 (12)^b	$H_{3,33}$ = 11.64	0.01
Capture success						
Dives/min.	1.66 ± 0.17 (15)^{ab}	0.75 ± 0.11 (8)^a	1.75 ± 0.37 (13)^{ab}	2.57 ± 0.40 (9)^b	$F_{3,41}$ = 4.95	0.01
Prey/min.	0.05 ± 0.04 (15)	0.26 ± 0.25 (8)	0.15 ± 0.10 (13)	0.22 ± 0.15 (9)	$F_{3,41}$ = 0.81	0.50

Entries in the same line that do not share the same letter were significantly different (Nemenyi test, $P \leq 0.05$ for foraging adults/transect and Tukey test, $P \leq 0.05$ for capture success).

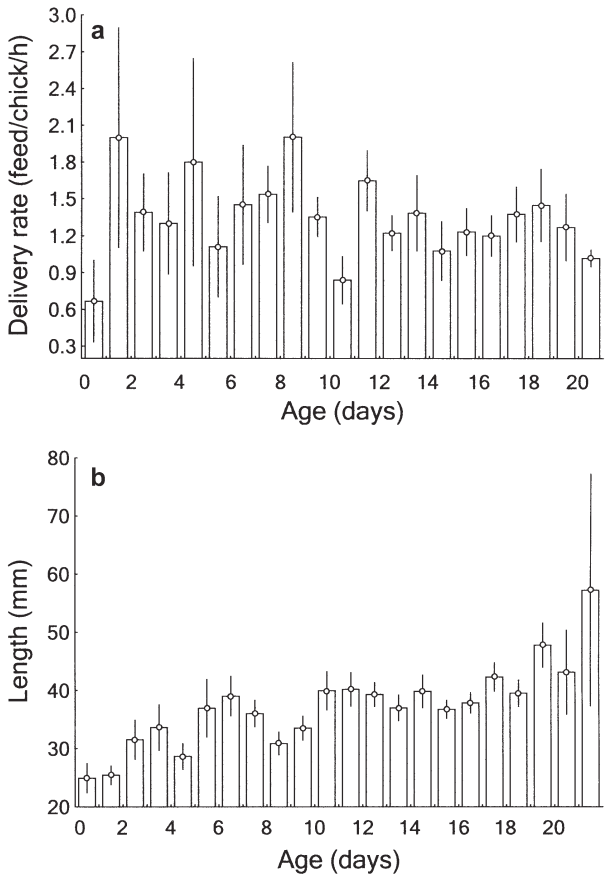


Figure 3. Relationship between: (a) prey delivery rate to *A* and *B* chicks (mean ± se) and chick age (n = 3–15, median = 9); and (b) prey length (mean ± se, n = 3–76, median = 35) and chick age.

The size of *Atherina* spp., *Fundulus* sp. and *Sardina pilchardus* delivered to chicks showed significant differences among tidal phases. Larger items were delivered during receding and incoming tides for *Atherina* spp. (Kruskal–Wallis: $H_{3, 164}$ = 22.52, P = 0.001), high and incoming tides for *Fundulus* sp. ($H_{3, 261}$ = 14.64, P = 0.002), and receding tide for *Sardina pilchardus* ($H_{3, 63}$ = 9.68, P = 0.02). Larger *Fundulus* sp. were delivered during intermediate and high wind speed (2–4 and >4 m/s; Kruskal–Wallis: $H_{2, 261}$ = 10.08, P = 0.01). For *Scomberesox saurus* larger prey items were delivered during intermediate wind speed (2–4 m/s; Kruskal–Wallis: $H_{2, 22}$ = 8.48, P = 0.01). In the case of *Atherina* spp., larger items were delivered during higher wind speed (>4 m/s; Kruskal–Wallis: $H_{2, 164}$ = 10.02, P = 0.01, Fig. 5).

Calorific content

Table 4 presents the energetic content of various prey items ingested by Little Tern chicks. *Atherina* spp. was significantly less calorific ($F_{3, 33}$ = 9.71, P = 0.001) than *Belone belone*, *Fundulus* sp. and *Sardina pilchardus*.

A and *B* chicks that received more than 30% of *Fundulus* sp. had a greater LGR (2.50 ± 0.15 g, n = 8) than those that were fed with more than 30% of *Atherina* spp. (2.31 ± 0.11 g, n = 11), although this difference was not significant (t_{17} = 1.06, P = 0.19). There was also little difference in LGR between the chicks whose diet had more than 14% of shrimps (2.21 ± 0.16 g, n = 7) and those that had less than 5% of shrimps (2.42 ± 0.14 g, n = 11; t_{16} = 0.96, P = 0.23). However, chicks that received more than 50% shrimps (less calorific prey) were fed at a higher rate (1.48 ± 0.27 fd/ch/h, n = 44; this includes two chicks of one nest that received 53.1% of shrimps) than those that received more than 50% *Fundulus* (more calorific items) (0.77 ± 0.11 fd/ch/h, n = 45; see also Table 4). This suggests that parents were able to compensate for the lower energetic value of shrimps.

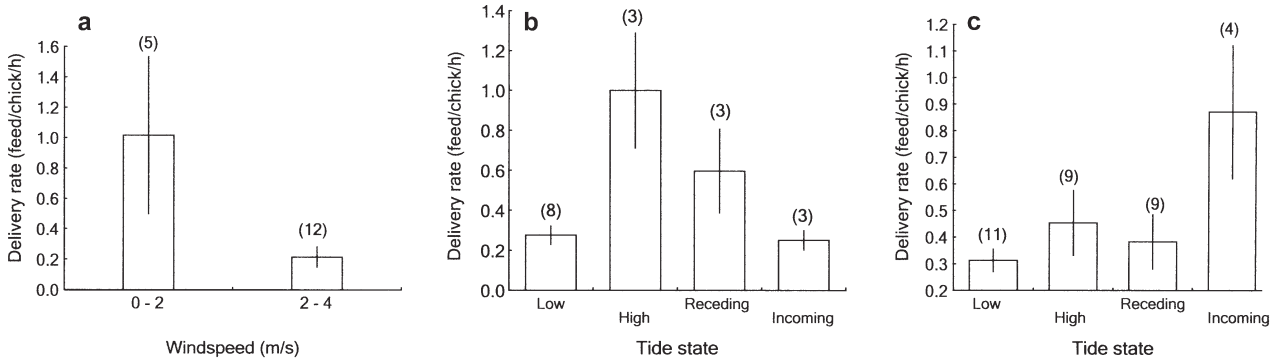


Figure 4. Significant relationships between wind speed and tidal phase on the prey delivery rate (mean ± se). Samples sizes are given above the columns. a, *Pomatoschistus* sp.; b, *Scomberesox saurus*; c, *Sardina pilchardus*.

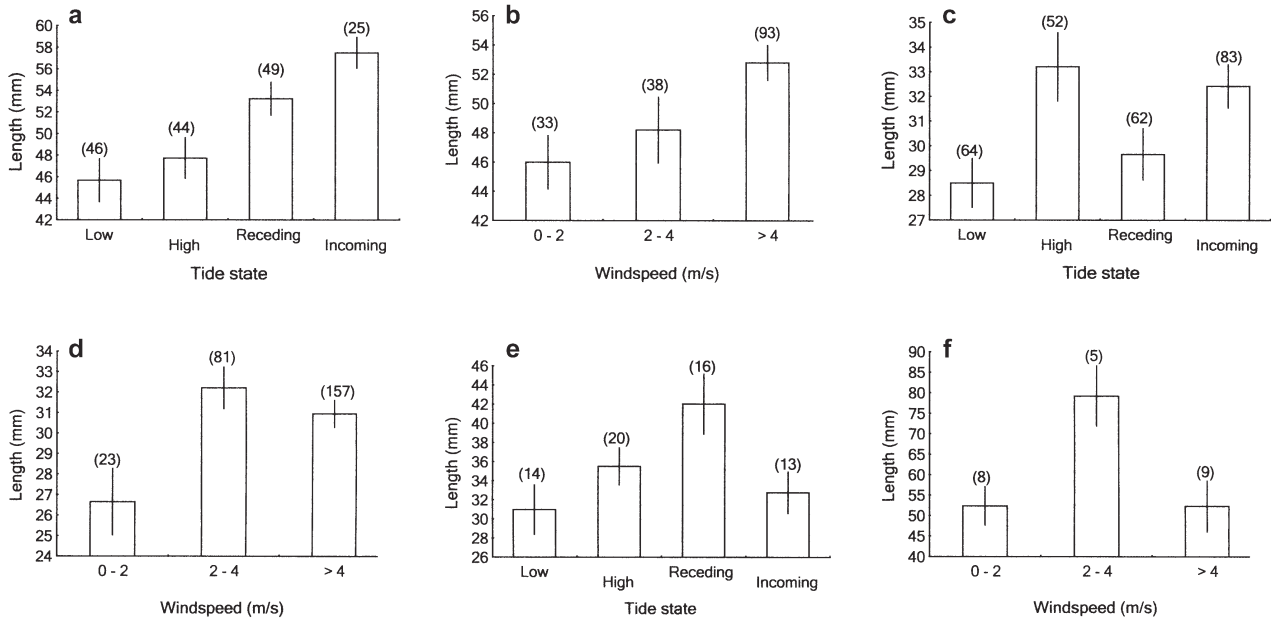


Figure 5. Significant relationships between wind speed and tidal phase on the length of delivered prey items (mean ± se). Samples sizes are given above the columns. a,b, *Atherina* sp.; c,d, *Fundulus* sp.; e, *Sardina pilchardus*; f, *Scomberesox saurus*.

DISCUSSION

Chick growth

The high mortality of C chicks has been well documented in Little Terns (Davies 1981, Norman 1992) as well as in other tern species such as the Common Tern *Sterna hirundo* (Nisbet *et al.* 1995, 1998). In general, this pattern may be explained by the insurance hypothesis. Adults with more experience invest initially in three chicks; if the food supply is good, parents can raise the three chicks, otherwise the last hatched chick often dies (Parsons 1970, O'Connor 1979).

Growth curves of fenced and unfenced *A* and *B* chicks were similar, suggesting that fences had no influence on chick growth. The differences registered in the early growth parameters between habitats, with higher growth rates on sandy beaches than on salinas, may be related to differences in chick diet or in the quality of their parents. Catry *et al.* (2004) showed that Little Terns breeding on sandy beaches in the Algarve laid larger eggs and clutches than those breeding on salinas, and suggested that these could be related to higher quality and/or older parents breeding on sandy beaches. Our growth data fit this hypothesis because, although there were some differences in chick diet between salinas and beaches, the most energetic prey item (*Fundulus* sp.) was delivered mostly to chicks on the salinas.

The mean wing-length growth rate of 6.16 mm/day was similar to that obtained by Norman (1992) in Gronant, Wales (6.80 mm/day). Also, some chick mass values in our study (6.30 g, 7.30 g, 13.23 g for 0, 1 and 4 days, respectively) are close to those obtained by Davies (1981) for the same ages (7.0 g, 9.8 g, 12.1 g) at Gibraltar Point, Lincolnshire. It should be noted that our study was in salinas but the studies of Norman (1992) and Davies (1981) were carried out on beaches in northern Europe without salinas.

Food delivery

We found a strong relationship between the species composition of delivered and dropped prey. Therefore, an insight into Little Tern chick diet in the Algarve, southern Portugal, may be obtained by collecting dropped prey items. This contrasts with data obtained by Bogliani *et al.* (1994) in the Comacchio lagoon, north Adriatic coast, where significant differences were detected between the percentage composition of dropped and ingested prey species. This difference is attributed to the availability of larger and bulkier species, such as Crucian Carp *Carassius carassius* in their study area. In our study, most prey species were longer and thinner. Of wider species such as Gobiidae, only small items were found both dropped and ingested.

Our study revealed strong differences between the diet

Table 4. Energy content, length and delivery rate (mean ± se, with *n* in parentheses) of prey items ingested by Little Tern chicks in the salinas of Santa Luzia, Algarve.

Species	Energy (kJ/g dry wt)	Length (mm)	Delivery rate (feeds/nest/hour)	Delivery rate (feeds/chick/hour)
Pelagic fishes				
Atherina spp.	18.48 ± 0.61 (22)	50.37 ± 0.95 (164)	0.91 ± 0.09 (75)	0.46 ± 0.04 (75)
Sardina pilchardus	20.48 ± 0.56 (8)	35.60 ± 1.32 (63)	0.58 ± 0.08 (29)	0.32 ± 0.05 (29)
Belone belone	20.32 ± 0.21 (11)	83.66 ± 1.97 (52)	0.46 ± 0.07 (34)	0.30 ± 0.04 (34)
Scomberosox saurus	19.22 (1)	58.44 ± 4.13 (22)	0.53 ± 0.10 (16)	0.30 ± 0.07 (16)
Mugilidae	20.35 (1)			
Euryhaline fish				
Fundulus sp.	21.95 ± 0.56 (8)	30.96 ± 0.53 (261)	1.29 ± 0.13 (72)	0.66 ± 0.08 (72)
Bottom/demersal fishes[1]		32.60 ± 1.58 (36)	0.38 ± 0.05 (21)	0.20 ± 0.03 (21)
Parablennius galerita	21.53 (1)			
Dicentrarchus punctatus	19.49 ± 0.09 (2)			
Coryphoblennius gallerita	22.28 (1)			
Diplodus vulgaris	20.26 (1)			
Symphodus melops	20.08 (1)			
Pomatochistus rispidus	19.26 ± 0.12 (3)	26.76 ± 0.86 (33)	0.64 ± 0.19 (17)	0.45 ± 0.17 (17)
Squid				
Loligo sp.	23.61 (1)			
Shrimps		25.61 ± 0.33 (150)	1.03 ± 0.14 (34)	0.71 ± 0.13 (34)
Paleomonetes varians	17.38 (1)			
Paleomon serratus	18.71 (1)			

[1]Length and delivery rate refer only to items of the families Blenniidae and Gobiidae.

of chicks in salinas (mainly *Fundulus* sp. and shrimp both in 2002 and 2003) and beaches (*Sardina pilchardus* and *Belone belone* in 2003), although *Atherina* spp. were important in both habitats. *Fundulus* sp., a main item of chicks in the salinas, is presumably captured in channels and salinas near the Santa Luzia breeding colonies because this is a euryhaline species not found in the main lagoon or the sea (Erzini *et al.* 2002). Another interesting aspect is that *Fundulus* sp. was not reported in the diet of chicks in 1989 by Calado (1989) at two different salinas of Ria Formosa. Presumably this fish species is a recent colonist in the area.

The diet of *A* and *B* chicks was similar, therefore the difference in growth rates between them was related to differences in their ingestion rates. The delivery rate was also related to the calorific content of prey, because chicks that received a higher percentage of shrimps were fed at a faster rate than those that received a higher percentage of fish. Similar observations were reported by Davies (1981) and Norman (1992) for Little Terns in Great Britain. The mean delivery rate of prey to Little Tern chicks in our study was similar to that obtained by Davies (1981) for a nest of three chicks at Gibraltar Point, Lincolnshire (0.90 fd/ch/h). In our study, mean delivery rate was not correlated with mean age of chicks. On the other hand, there was an increase in mean length of prey delivered to Little Tern chicks as they grew. This suggests that either prey were increasing in size or parents selected larger prey items for older chicks. Our study gives more support to the second hypothesis because there was a steady increase in the length of fish delivered as chicks grew. Therefore, parents met the increasing energy demand of their growing chicks by adjusting prey size, rather than prey frequency. Similar results were found by Bogliani *et al.* (1994) for Little Terns and Stienen *et al.* (2000) for Sandwich Terns *Sterna sandvicensis*. Since young chicks cannot swallow larger prey, the availability of smaller prey could determine the timing and success of breeding (Safina *et al.* 1988).

Massias & Becker (1990) argued that shrimps are an alternative energetic resource for Common Tern chicks in periods of absence of more energy-rich species. Our study agrees with this because chicks fed mostly on shrimps, which had the lowest calorific content, were fed at a higher rate. We also observed shrimps being captured in salinas 10–50 m away from the Santa Luzia breeding colonies, which should contribute to lowering the energy expenditure in foraging by adults breeding in salinas.

Effects of tidal phase and wind speed on the delivery of specific prey species

Although a greater number of adults were observed foraging in the lagoon habitat at low tide (both in 2002 and 2003), tide had no influence on prey delivery rate to chicks. This could be explained if adults use low tide to fulfil their daily energy requirements. However, adults dived more during incoming and receding tides, presumably when most prey are moving in the water column. We suspect that low tide constrained the availability of estuarine prey species, as suggested by the low capture success that we registered during this tidal phase. Stienen *et al.* (2000) observed also that the success of Sandwich Terns foraging in the Wadden Sea was low during low tide. On the contrary, salinas and channels around the colonies should be more consistent in prey availability, because they are not subjected to tidal influences. A salina is composed of several tanks separated by soil banks that protect against strong winds, thus providing a calmer water surface and good visibility for diving. This may explain the delivery of larger *Fundulus* sp. individuals during high and incoming tides (when fewer adults are foraging in the lagoon habitat) and during adverse foraging conditions in the lagoon (Brenninkmeijer *et al.* 2002) and in the sea, such as during windier periods (Stienen *et al.* 2000).

Atherina is a resident lagoon species (Muzavor *et al.* 1993, Erzini *et al.* 2002), and normally the older (and larger) individuals live at greater depths. When tide varies between high and low, they move in the water column and become more accessible to Little Tern adults. This could explain the delivery of larger items during the incoming and receding tidal phases. *Atherina* spp. may also be more active during periods of greater wind speed, because the mixing of sea-surface waters may influence the cycle of nutrients in the water column. *Pomatoschistus* sp. is a benthic resident lagoon species (Muzavor *et al.* 1993, Erzini *et al.* 2002), which explains why their delivery rate was higher in calm conditions. For *Scomberesox saurus*, a marine species (Whitehead *et al.* 1984), larger prey items were delivered during intermediate wind speed periods, which are considered to be optimal conditions for tern foraging in the sea (Cramp 1985, Stienen *et al.* 2000). On the other hand, there was a greater delivery rate of this prey species during high and receding tidal phases, when we observed fewer adults foraging in the lagoon habitat.

In summary, Little Terns in the Algarve appear to rely on several species for rearing chicks. Although one species, *Atherina* sp., was the main prey species in both 2002 and 2003, there were important variations in prey species between habitats as well as in the importance of pelagic and bottom/demersal fishes between 2002 and 2003. The availability of prey species for the terns depends also on environmental conditions such as wind speed and tide that affect the availability of certain prey and/or the fishing success of the parents. The higher energy content of *Fundulus* sp. and the abundance of shrimps on salinas may be important criteria for Little Terns to breed there. Nevertheless, Little Tern parents experienced no difficulties in providing sufficient food to rear two chicks in both salinas and sandy beaches in the Algarve. Disturbance, predation and flooding (Catry *et al.* 2004) are usually more important in explaining breeding failure than food availability.

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ENDNOTE

a. $M_d = M_0 + ad + bd^2$, where M_d = chick mass on day d, M_0 = estimated chick mass on day 0, d = age in days, a = linear growth parameter, the instantaneous slope of the growth curve on day 0, and b = quadratic growth parameter (a measure of the curvature of the growth curve; Nisbet *et al.* 1998). M_0, a and b were estimated for each fenced and unfenced chick for which mass was measured on day 0 and at least two other days

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Kentish Flats Offshore Wind Farm: Diver Surveys 2009-10



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Executive Summary

Surveys of the distribution and abundance of red-throated divers at the Kentish Flats wind farm during the pre-construction, construction and post-construction phases of the wind farm have been undertaken over 8 years prior to 2009-10. This report presents an analysis of new data sets from surveys completed at Kentish Flats during 2009-10.

The original statistical analysis carried out for the monitoring conducted under the FEPA licence of the effects of the Kentish Flats wind farm on bird populations utilised a BACI (before-after-control-impact) approach. However it was acknowledged in the monitoring reports that this was of relatively low power which meant that no statistically significant effect was identified although a qualitative assessment suggested an avoidance of the site by divers.

A previous review of the Kentish Flats monitoring programme data and re-analysis had shown that there has clearly been a notable decline in diver numbers within the wind farm/buffer zone following construction, at a time when the wider population (as determined from the aerial surveys) appeared to have been relatively stable, confirming the qualitative observations made in the previous Kentish Flats monitoring reports. There had been a statistically significant decrease in diver numbers within the wind farm site and its surrounds. There has not only been a decrease in numbers but also a shift in distribution away from the wind turbines, most markedly within 500m. This reduction in the wind farm site and 500m buffer was also apparent from the 2009-10 data. As noted in the previous report, the biological importance of such behaviour is not yet clear and needs to be addressed with reference to the context of the wider diver population within the Outer Thames Estuary.

There was some suggestion from the 2008-09 data in particular that the magnitude of the displacement may be decreasing through time; divers may be habituating to the presence of the wind turbines. The 2009-10 data did not suggest that the strength of any such habituation was increasing, though there were more diver records from within the wind farm site than there had been in 2008-09.

As noted previously, it is important to recognise that the results for this site may not be directly applicable to other wind farm sites given the Kentish Flats wind farm's relatively small number of turbines and footprint size, and its relatively low importance for divers. It is possible, for example, that divers using a site of greater importance/attractiveness to them may be less likely to be affected by disturbance than those at Kentish Flats.

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1. Introduction

Background

Bird survey work for the Kentish Flats offshore wind farm was carried out between 2001 and 2007 using both boat based and aerial survey methods. Initially this provided baseline data for the project impact assessment and Environmental Statement [ES]. Subsequently a FEPA monitoring program through the pre-construction (2002 – 2003) and construction phase (August 2004 – August 2005) and for 3 years post-construction has been completed. The final ornithological monitoring report was produced by Environmentally Sustainable Systems Ltd [ESS], the project lead ornithological consultants, in July 2008 (Gill et al. 2008) which reported on the findings of the three year post-construction monitoring program.

The key ornithological issue identified at the Kentish Flats as a result of the FEPA monitoring is a possible effect on divers during the operational phase. The Thames Estuary as a whole has recently been shown to be of major international importance to this group, particularly red-throated divers, with a wintering population estimated at about 8,000 birds (O'Brien et al. 2008). The conclusion of the ESS monitoring report with respect to divers was that there was no evidence of any statistically significant effects of the wind farm on divers (when comparing the wind farm with the available data for the control site), although they did note an apparent displacement of divers from the operational turbine array based on a qualitative review and observations reported by the bird surveyors.

The bird monitoring required under the FEPA licence issued for the Kentish Flats wind farm site came to an end in 2008 and the final bird monitoring report has been accepted by Natural England and the Marine Fisheries Agency (MFA) so that no further statutory requirement for monitoring exists at the site. However, in recognition of the observations relating to the apparent avoidance of the turbine area by divers, Vattenfall (the owners and operators of the Kentish Flats site) have decided to undertake further, focused boat based surveys during the winter of 2008/2009 (Percival 2009) and 2009/10 on a voluntary basis to further investigate this issue. This report covers the second of those winter periods.

The original statistical analysis of the effects of the Kentish Flats wind farm on bird populations utilised a BACI (before-after-control-impact) approach. However it was acknowledged in the monitoring reports that this was of relatively low power which meant that no statistically significant effect was identified.

Objectives

The current study has allowed the application of a more powerful statistical approach using the specific recorded locations of each diver observation. Diver densities recorded in zones within and around the wind farm and their distances from wind turbine locations (before/during and after construction) have been analysed. Although limited by the extent of the survey area and the lower coverage of the control area, this approach has enabled a revised statistical analysis to be undertaken and updated conclusions to be reached that accord

better with the observations made by field observers and reported in the previous monitoring reports – for a detailed account of the revised analysis of the FEPA data see Percival (2009).

The data re-analysis has shown that there has clearly been a notable decline in diver numbers within the wind farm/buffer zone following construction, at a time when the wider population (as determined from the aerial surveys) appears to have been relatively stable (as was that in the Control area, albeit with a limited data set). This confirms the qualitative observations made in the previous Kentish Flats monitoring reports. There has been a statistically significant decrease in diver numbers within the wind farm site and its surrounds. There has not only been a decrease in numbers but also a shift in distribution away from the wind turbines, most markedly within 500m. The current report provides additional field data from another winter period to test this result further, to examine the patterns of diver distribution and behaviour in relation to the operational wind farm.

There was some suggestion from previous analysis that the magnitude of the displacement may be decreasing through time; divers may be starting to habituate to the presence of the wind turbines. Another objective of the 2009-10 work was therefore to obtain further data to test this hypothesis.

2. Survey Methodology

The survey methodology for 2009-10 was the same as that followed in 2008-09, though gulls were again recorded (as in the year prior to 2008-09) and no vantage point survey were carried out (it had been concluded in the 2008-09 report that these would not be likely to add much to our knowledge of the interaction of the divers and the wind farm given the very low numbers observed flying through the wind farm; Percival 2009).

The same surveyors and survey vessel were used as previously, but the survey methodology was tailored to obtain better data on the key issue, behaviour of divers, whilst at the same time maintaining compatibility/comparability with the previous longer-term data set. This also allowed testing of some of the assumptions made previously and provided a better estimate of diver numbers in the study area.

The key differences in the survey method used in 2009-10 to those used prior to 2008-09 were as follows:

- The survey scan was extended from 90 to 180 degrees, recording on both sides of the boat. This effectively doubles the survey area.
- Divers more than 480m ahead of the boat were recorded and the distance at which they flushed estimated. This was to enable a correction to be applied to the previous data to account for this flushing effect.

All of these improvements maintain backwards-compatibility with previous surveys to allow for analysis of long-term changes.

For consistency the same survey transects were used as previously for the wind farm/buffer and control areas. These are approximately 1km apart, rather than the more usual 2km separation, which means that the potential for double-recording of mobile species such as divers is more likely. The absolute population estimates presented here should therefore be treated with caution as they may over-estimate the actual numbers. Particular care should be taken comparing these numbers with those from surveys using a wider transect separation. However, for the purposes of this analysis it has been assumed that this has not led to any systematic bias in the data set.

In addition the survey area was extended in 2009-10 to cover a wider area around the wind farm site, primarily to provide data for a possible extension to the wind farm. The extent of the current and previous survey areas and the current and previous survey transects are shown in Figure 1. The results for that additional area are shown in the bird distribution maps but are not included in any of the main population estimates, to maintain consistency and comparability with previous surveys. Population estimates for that area are presented separately.

A total of six surveys were carried out during November-February 2009-10. The survey dates and coverage were as follows:

- 30/11/09 – wind farm plus buffer and extension area.
- 16/12/09 - wind farm plus buffer, control and extension area.
- 11/1/10 - wind farm plus buffer, control and extension area.

- 19/1/10 - wind farm plus buffer, control and extension area.
- 3/2/10 - wind farm plus buffer, control and extension area.
- 19/2/10 - wind farm plus buffer, control and extension area.

3. Data Analysis

The additional analysis was carried out following the same analytical strategy as in 2008-09. This sought to address the specific questions of:

- How have diver numbers changed following construction of the wind farm?
- How has diver distribution changed and is this continuing to change through time?

One key principle in these analyses is to maximise the use of the information contained within the raw data, for example by using raw diver locations rather than just transect summaries.

5.1 Diver distribution and abundance 2009-10

The diver population estimates within the wind farm buffer and control areas based on the 2009-10 survey data are summarised in Figure 2. This Figure shows the estimated diver population for each survey for the wind farm buffer and control areas. Numbers in both these areas were highly variable through the winter period, as found in previous years. As previously these numbers have been calculated using correction to allow for survey coverage and for declining detectability of birds at increasing distance from the survey vessel (applying distance correction factors from the London Array wind farm baseline surveys – see Appendix 1).

Figure 2 also shows the population estimate for the new extension to the study area surveyed in 2009-10 for the first time. It held similar numbers to the control area (mean 22, peak 57).

As found previously, a very high proportion (98%) of all of the divers that were identified to species in 2009-10 were red-throated divers. The remainder were black-throated. Previously there had been an increase noted in the proportion of black-throated diver towards the end of the winter period, but in 2009-10 all records of this species were from the earlier part of the winter (with none after 10/1/10).

The diver distribution maps from the 2009-10 boat surveys are shown in Figure 3a. This map plots out all of the raw sightings, so when interpreting them it is important to consider that the control area was surveyed for one visit less than the rest of the study area. Diver distribution maps for the previous survey periods are given for comparison in Figures 3b-e.

5.2 Effects of the Wind Farm on Diver Distribution and Abundance

As in the previous report (Percival 2009) two key parameters have been used in the analysis of the effects of the wind farm on the divers; the diver populations in

buffer zones around the wind farm and the distance of diver records from the wind turbine locations.

Diver Population Estimates

The mean and peak diver counts through the main diver period (Nov-Mar) are summarised in Table 1 for the wind farm/buffer area and for the control area. There has been a marked drop in diver numbers in the wind farm/buffer study area since the construction of the wind farm and this has continued into 2009-10. Data for the western Thames Estuary aerial survey area (TH1) are also given for comparison where these are available, to give a wider context of any population changes (though taking into account that a small part of the survey area - i.e. that in which the wind farm is located - was not surveyed). Some caution is required in directly comparing the population densities derived from the aerial and boat surveys: given experience from other surveys in the Outer Thames Estuary since it is likely that the aerial surveys under-estimate the actual numbers of divers (Percival et al. 2005).

Table 1.Mean and peak diver population estimates for the wind farm/buffer, control area and wider aerial survey area, 2002-2010.

Winter	Phase	Boat survey: Wind farm + buffer		Boat survey: Control		Aerial Survey TH1	
		Mean	Peak	Mean	Peak	Mean	Peak
2002-03	Pre	608	2,226	47	196	3,340	5,227
2003-04	Pre	552	1,313	5	19	1,214	1,644
2004-05	Construction	945	2,039	17	58	1,934	3,166
2005-06	Post	119	408	17	64	1,961	2,725
2006-07	Post	136	317	15	60	(2,009) *	(2,009) *
2008-09	Post	86	171	186	646	No data	No data
2009-10	Post	72	187	17	54	No data	No data

** In 2006-07 aerial surveys were only carried out during February 2007.*

There has clearly been a notable decline in diver numbers within the wind farm/buffer zone following construction (Kruskal-Wallis non-parametric ANOVA: H=8.4, p=0.015, 2df), at a time when the wider population (as determined from the aerial surveys) appears to have been relatively stable. This trend continued in 2009-10, adding further confirmation to the qualitative observations made in the Kentish Flats monitoring reports prior to the 2008-09 season and to the analysis presented in the 2008-09 report (Percival 2009). The higher numbers recorded in the Control area in 2008-09 were not observed in 2009-10 (when numbers returned to their more usual lower level).

Diver Distribution in relation to the Wind Farm

Looking in more detail at the spatial pattern of those changes, the following zones were used: wind farm footprint and buffers of 500m, 1km, 2km and 3km. Figure 4 shows the population estimates in each of these zones during the pre-

construction, construction and post-construction periods, updated to include the 2009-10 results. Figure 5 shows the equivalent analysis for population densities (i.e. standardised for the area of each zone).

Figures 4 and 5 both show that the diver populations dropped markedly in the wind farm footprint and the 500m buffer around that in the construction year, with an increase in numbers in the 2-3km buffer, suggesting that there had been some displacement from the wind farm. The post-construction data showed a marked decline, and that decline extended over all of the study area, including in 2009-10. The diver numbers in the whole study area have clearly been lower during the post-construction surveys. That decline has been apparent across the study area but have been most marked in the zone within 500m of the wind turbines.

Table 2 gives the diver densities (population estimates standardised for the area in each buffer zone to give a mean density per km^2) observed in each of the buffer zones and the proportionate change in diver density observed in each during the pre-construction, construction and post-construction periods. Densities during the construction phase declined only in the zones within 500m of the wind turbines, with higher numbers recorded in the 2-3km zone. Densities recorded during the post-construction surveys have been rather lower across the whole survey area, though with the magnitude of that decline higher in closer proximity to the wind farm.

Table 2. Diver densities (number per km^2) in each of the wind farm buffer zones during the pre-construction, construction and post-construction periods, and the percentage change from the pre-construction baseline.

Period	Wind farm	0-0.5km	0.5-1km	1-2km	2-3km
Pre-construction	3.5	5.1	8.4	7.7	9.8
Construction	1.1	0.8	9.2	3.3	48.0
Post-construction	0.2	0.7	2.0	3.0	3.6
% change from pre-construction baseline:					
Construction	-69%	-84%	+10%	-57%	+488%
Post-construction	-95%	-87%	-76%	-61%	-63%

Diver Proportionate Distribution in relation to the Wind Farm

An alternative way to explore the effects of the wind farm is to look at the proportionate distribution of birds across these zones. This takes into account differences in overall numbers between years by analysing the proportion of divers records from each of the buffer zones rather than the counts/densities. For each zone the population estimates are expressed as a proportion of all of the records from within the study area. Figure 4 shows the proportion of diver records in each of these zones during the pre-construction, construction and post-construction periods. This highlights the decline in diver activity within 500m of the wind farm, which has continued in 2009-10. The 'expected' proportion in each zone is also shown on the Figure (as a dashed red line) if the distribution were uniform across the whole study area (i.e. if the number of diver records in each zone was proportional to the area of that zone surveyed). The proportionate

distribution of the divers in relation to the wind farm was similar to that recorded in the previous post-construction surveys, with only 2% of records observed within the wind farm. The wind farm site and the 500m buffer around it were both under-utilised throughout the surveys, including those carried out prior to construction,. There was a small increase in use of the wind farm site in 2009-10 over 2008-09, when there was only a single record of a diver from within the wind farm – 5 were seen there in 2009-10 (three singles and a flock of 2). This may be an indication that the divers are becoming more tolerant of the wind turbines. Overall the diver distribution remains significantly different from both a uniform and the pre-construction baseline (X^2 = 22.7 and 22.2 for the uniform and pre-construction comparisons, p <0.001 in both cases).

Table 3 summarises the diver proportionate distributions observed in each of the buffer zones and the percentage change in those proportions observed in each during the pre-construction, construction and post-construction periods. The percentage of diver records from within the wind farm has declined from 9% pre-construction to only 2% during construction and post-construction. This represents a reduction by 79% from the pre-construction value during construction and by 81% from the pre-construction value during the post-construction period. Substantial percentage declines were also recorded in the 0-0.5km buffer zone around the wind farm (10% of records prior to construction, 1% and 4% during construction and post-construction respectively). This represents a reduction by 89% of the pre-construction value during construction and by 59% from the pre-construction value during the post-construction period. The changes more than 500m from the wind farm are much lower, though the results do suggest a smaller-scale reduction up to 1km from the wind farm (where the percentage of diver records has dropped by 24% of the pre-construction baseline during construction and by 21% post-construction). Beyond 1km no decline in the proportionate distribution was noted during the post-construction period.

These changes in the proportionate distribution are considered here to be more robust than the changes in density (Table 2) as they are less sensitive to changes in the overall diver numbers in the region. However the limited spatial scale of the boat survey area at Kentish Flats means that caution needs to be applied in interpreting these values and in applying them elsewhere.

Table 3. The proportions of diver records in each of the wind farm buffer zones during the pre-construction, construction and post-construction periods, and the percentage change in those proportions from the pre-construction baseline.

Period	Wind farm	0-0.5km	0.5-1km	1-2km	2-3km
Pre-construction	9%	10%	21%	41%	18%
Construction	2%	1%	16%	12%	69%
Post-construction	2%	4%	17%	54%	24%
% change from pre-construction baseline:					
Construction	-79%%	-89%	-24%	-70%	+372%
Post-construction	-81%	-59%	-21%	+30%	+29%

Clearly there has been a statistically significant decrease in diver numbers within the wind farm site and its surrounds, and this has continued in 2009-10. There has not only been a decrease in numbers but also a shift in distribution away from the wind turbines, most markedly within 500m.

Spatial Distribution of Divers in relation to Distance from Wind Turbines

Additional analysis was also undertaken of the spatial distribution of diver records in relation to distance from the wind turbines. If the birds were displaced then one would expect that distance to increase after the turbines had been constructed. For this analysis the control area data have been excluded due to the reduced coverage of the surveys in that area and so that the analysis focussed more closely on the area in proximity to the wind farm. The results summarised in Table 4 show that divers were found significantly further from the turbines during the construction and post-construction surveys (ANOVA $F=14.9$, $p_{4,1219}<0.001$). Pairwise comparisons showed that the post-construction diver distances from turbines were significantly greater than pre-construction for all of the years compared. Comparing the construction and post-construction years, the lowest distances were recorded in the construction year but those in the most recent survey year, 2009-10, were not significantly different from the construction year. In the intervening post-construction years divers were recorded at significantly greater distances from the turbines than in either the construction year or 2009-10. The mean distance to a series of random points is also given for comparison. These results add further support to the conclusion that displacement of red-throated divers has occurred from the wind farm site to a distance of about 500m.

Table 4. Distances between diver records and the wind turbine locations during the pre-construction, construction and post-construction phases.

Period	Mean distance from turbines (km)	Standard error	95% confidence limits
Pre-construction	1.18	0.03	1.12-1.24
Construction	1.38	0.06	1.27-1.49
Post-construction (2005-07)	1.53	0.05	1.43-1.62
Post-construction (2008-09)	1.63	0.07	1.49-1.77
Post-construction (2009-10)	1.44	0.06	1.33-1.55
Random points	1.09	0.04	1.01-1.17

The mean distances from the turbines recorded in each winter are shown in Figure 5 to illustrate the between-year differences. The mean distance was significantly smaller small in both of the pre-construction years in comparison with the construction and post-construction years. The 2009-10 data showed a reduction in the mean distance between the divers in comparison with the

previous year, a possible indication of increasing tolerance of the turbinesbut were still statistically significantly greater than the pre-construction years.

4. Conclusions

The analysis of the 2009-10 data collected at Kentish Flats has shown that the decrease in diver numbers within the wind farm site and its surrounds (particularly within 500m of the turbines) has continued into the fifth winter since construction.

There have been some indications of an increased use of the area in proximity to the wind turbines compared with previous post-construction years (particularly in 2008-09) and this may indicate that the divers are starting to habituate to the presence of the wind turbines. However use of the wind farm site by divers continues to be very low, though even in the pre-construction baseline years use of the wind farm site was still relatively low. As noted in the previous report, the biological importance of such behaviour is not yet clear and needs to be addressed with reference to the context of the wider diver population within the Outer Thames Estuary (particularly the Special Protection Area).

An important question with regard to the local effect of the Kentish Flats wind farm on divers (displacing divers from the wind farm site and its surrounds) is its context in the wider Outer Thames Estuary SPA diver population. The area affected is very small as a proportion of the area used by divers (the wind farm plus a 500m buffer occupies only 0.004% of the Outer Thames SPA), and the aerial survey results suggest that Kentish Flats is not particularly important for divers (with a raw observed density of 1.5 divers per km^2 in that area, compared with densities in excess of 10-fold this amount in more preferred parts of the aerial survey area). This displacement effect is therefore probably negligible in the context of the Outer Thames diver population as a whole but further investigation would be needed to test this hypothesis. Further monitoring data from other offshore wind farms should also provide useful data to address this question, though methods for that monitoring should be tailored to obtain the best possible data on divers.

Whilst these results continue to show a clear reduction in diver numbers within the Kentish Flats wind farm, it was apparent that the magnitude of that reduction appeared to be reduced from the 2008-09 surveys. This effect was less clear from the 2009-10 data but may continue through time as has been observed at some onshore wind farms (e.g. for pink-footed geese, Madsen and Boertmann 2008). Studies of divers at existing offshore wind farms have also reported displacement from the wind farms. At Horns Rev none were found within the wind farm after construction, during a three-year monitoring programme (Petersen et al. 2006) and a reduction in density was reported up to 2km from the wind farm. The same publication also reported a reduced diver utilisation of areas within 2km of the Nysted wind farm after construction. Post-construction monitoring at the Nordzee wind farm project (Leopold et al. 2010) has not found any significant effects on divers, though those wind farms are located outside the divers' main preferred areas.

As noted in the previous report, it is also important to recognise that the results for this site may not be directly applicable to other wind farm sites given the Kentish Flats wind farm's relatively small number of turbines and footprint size, and its relatively low importance for divers. Divers using a site of greater importance/attractiveness to them may be less likely to be affected by disturbance than those at Kentish Flats. Differences in susceptibility to disturbance in relation to resource availability have been noted in other bird-wind farm interactions. For example, foraging barnacle geese have been reported as being displaced from as far as 600m from wind turbines on farmland habitat in winter (Kowallik and Borbach-Jaene 2001) yet birds from the same population feed as close as 25m to turbines during spring staging on Gotland (Percival 1998), where the birds are feeding on a much scarcer and more nutritionally valuable saltmarsh habitat in proximity to wind turbines. Displacement from less preferred feeding sites may more readily occur than from more important foraging areas (where birds may be more tolerant of the presence of the wind turbines).

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APPENDIX ONE – Distance correction factors and calculations. [Originally produced as Appendix 5 to the London Array Wind Farm ES]

As noted in the main ES, the raw count data from both the boat-based and the aerial surveys need to be adjusted to take into account the fact that the likelihood of a bird being seen declines with distance from the observer (i.e. detectability is a function of distance from the transect line). Put simply, the chance of seeing a bird close to the observer would be higher than if it were at greater distance. The relationship between detectability and distance can be modelled using software packages such as Distance (Buckland *et al*. 2001), but for the purposes of this assessment a simpler approach was adopted (mainly because the limited number of distance bands makes modelling of the distance function difficult for many of the species encountered in this study). The approach used here is similar to that used by JNCC in their Seabirds at Sea surveys (e.g. Stone et al. 1995), but correction factors have been calculated for each major species group (divers, seaduck, gannet, gulls, terns and auks) specifically using the data collected from each of the two survey methods (boat and aerial). Species were assigned to these groups on their similarly of likely detectability and pooled to give a robust sample size for each group. Group compositions are given in Table A3.1. The same process was used to correct both the aerial and the boat data, though as detectability differed between these methods separate correction factors were calculated for each.

Table A3.1. Species groups used in calculation of distance correction factors

Species Group	Species
Divers	Divers, cormorants.
Gannet	Gannet
Gulls	Gulls, skuas, terns, shearwaters
Seaduck	All wildfowl

The process in calculating those correction factors was as follows:

- The total numbers of birds of each species group were calculated for each distance band over all of the surveys.
- Differences in the width of the distance bands were taken into account by dividing the total number by the band width, to give a standardised total (density index).
- It was assumed that bird detectability in the closest transect to the observer was 100% (a standard assumption of the Distance sampling methodology).
- As detectability of birds on the sea and flying were different from the boat survey data separate correction factors were used for each of these. In fact detectability of flying birds was so high that no correction factors were necessary for these birds – effectively all of these birds were detected within the main transect.
- For each of the other bands, the percentage difference between that band's standardised total and the closest band to the observer were calculated.

- These differences were then applied as the correction factors, dividing each count by the appropriate factor. For example, auks in band C were divided by 41%. Hence a count of 100 in that band would be corrected to 244 (=100/0.41).

Table A3.2. Distance correction factors used for the boat survey data

Species group	A [0-50m]	B [50-100m]	C [100-200m]	D [200-300m]	E [>300m] – out of transect
Auks	100%	100%	41%	21%	3%
Divers	100%	100%	90%	90%	60%
Gannet	100%	100%	100%	100%	100%
Gulls	100%	100%	76%	76%	74%
Seaduck	100%	100%	85%	85%	85%

Note: values are given in the Table for band E but these were not used in the density calculations or population estimates.

Table A3.3. Distance correction factors used for the aerial survey data

Species group	A [49-174m]	B [175-459m]	C [>460m]
Auks	100%	29%	2%
Divers	100%	42%	7%
Gannet	100%	62%	15%
Gulls	100%	40%	10%
Seaduck	100%	90%	21%

Note: values are given in the Table for band C but these were not used in the density calculations or population estimates.

APPENDIX TWO – 2009-10 BOAT SURVEY DIVER DATA.

Date	Transect	Time	Species	Flock	Distance Band	Direction	Fly	On Sea	Feed	Height	In Transect	Notes	Side	Latitude	Longitude
30/11/2009	Transect 0	09:02	BV	1	C			1				Flushed but struggled to get airbourne - full of fish?	S	51.45645	1.017273
30/11/2009	Transect 0	09:03	RH	1	D	SE	1			1			P	51.45653	1.024002
30/11/2009	Transect 1	09:41	RH	1	B			1				Flushed.	S	51.44415	1.053775
30/11/2009	Transect 2	10:25	RH	1	C	SW	1			35 1			P	51.45008	1.065
30/11/2009	Transect 3	11:14	RH	1	A	W	1			10			S	51.46692	1.059796
30/11/2009	Transect 6	12:55	RH	1	C	W	1			5			S	51.4856	1.081757
30/11/2009	Transect 6	13:23	RH	1	B	E	1			2 1			P	51.43585	1.155349
30/11/2009	Transect 7	14:05	RH	1	A			1				Flushed	S	51.48548	1.107185
30/11/2009	Transect 8	14:19	RH	1	E	W	1			15			S	51.47192	1.127991
30/11/2009	Transect 8	14:25	RH	1	C	N	1			2			P	51.46353	1.149498
16/12/2009	Transect 0	13:33	RH	1	B			1					S	51.43278	1.048981
16/12/2009	Transect 0	13:34	RH	3	A	W	1			2 1			S	51.43404	1.04644
16/12/2009	Transect 0	13:35	RH	1	D	SW	1			8			S	51.43687	1.046985
16/12/2009	Transect 0	13:47	RH	8	A	W	1			5			S	51.45417	1.021621
16/12/2009	Transect 0	13:47	RH	2	A	NE	1			2 1			S	51.45417	1.021621
16/12/2009	Transect 0	13:51	RH	3	A	W	1			3			P	51.46002	1.013248
16/12/2009	Transect 0	13:52	RH	7	A	W	1			2			S	51.46181	1.012036
16/12/2009	Transect 0	13:52	RH	3	D	N	1			10			S	51.46297	1.014698
16/12/2009	Transect 0	13:54	RH	1	A	W	1			2			S	51.46492	1.008609
16/12/2009	Transect 1	14:01	RH	1	A	E	1			15			S	51.47539	1.011949
16/12/2009	Transect 1	14:09	RH	1	C	E	1			5	1		P	51.46158	1.031865
16/12/2009	Transect 1	14:16	RH	2	A	E	1			10			S	51.44812	1.047112
16/12/2009	Transect 2	14:43	RH	2	D	W	1			1			S	51.43825	1.083474
16/12/2009	Transect 2	14:54	RH	1	D			1				flushed	S	51.45449	
16/12/2009	Transect 2	14:58	RH	1	C			1				flushed	S	51.46042	1.051185
16/12/2009	Transect 2	15:07	RH	2	D	W	1			3			P	51.4724	1.028885
16/12/2009	Transect 2	15:09	RH	2	A	W	1			2			P	51.47673	1.027642
16/12/2009	Transect 2	15:11	RH	2	D	W	1			10 1			S	51.48146	1.026897
16/12/2009	Transect 3	12:35	RH	1	C			1		1		flushed	P	51.4799	1.045428
16/12/2009	Transect 3	12:50	RH	1	C			1				flying along edge of array	S	51.45279	1.075731
16/12/2009	Transect 3	13:05	RH	1	D	E	1			5			S	51.42668	1.107717
16/12/2009	Transect 4	11:56	RH	1	A	E	1			10			P	51.43117	1.125574
16/12/2009	Transect 4	12:22	RH	1	D	SW	1			3 1			P	51.4742	1.063054
16/12/2009	Transect 4	12:23	RH	6	C	W	1			5 1			S	51.47823	1.065076
16/12/2009	Transect 4	12:23	RH	1	A	NW	1			10			S	51.47749	1.063713
16/12/2009	Transect 4	12:26	RH	2	C	S	1			12			P	51.48121	1.054871
16/12/2009	Transect 4	12:27	RH	1	B			1					S	51.48415	1.05487
16/12/2009	Transect 5	11:17	RH	1	B			1				flushed	S	51.47821	1.077582
16/12/2009	Transect 5	11:25	RH	1	B	E	1			2	1	flying through array	P	51.4674	1.095632

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1

Date	Transect	Time	Species	Flock	Distance Band	Direction	Fly	On Sea	Feed	Height	In Transect	Notes	Side	Latitude	Longitude
16/12/2009	Transect 6	10:37	RH	1	A	E	1			10	1		P	51.44507	1.140426
16/12/2009	Transect 6	10:54	RH	1	C	SE	1			15	1		P	51.47616	1.095149
16/12/2009	Transect 6	10:54	RH	2	C			1				flushed	S	51.47796	1.098371
16/12/2009	Transect 6	10:57	RH	1	C			1				flushed	P	51.48203	1.087734
16/12/2009	Transect 6	10:57	RH	2	C			1				flushed	S	51.48357	1.091286
16/12/2009	Transect 6	10:58	RH	3	C			1				flushed	P	51.48404	1.085483
16/12/2009	Transect 7	09:57	RH	1	B	NE	1			8[1]			S	51.48096	1.111054
16/12/2009	Transect 7	09:58	RH	1	E	SE	1			5[1]			P	51.48259	1.11952
16/12/2009	Transect 7	09:58	RH	3	A	E	1			10[1]			S	51.47974	1.113461
16/12/2009	Transect 7	10:00	RH	2	E	SE	1			10[1]			P	51.47964	1.12322
16/12/2009	Transect 7	10:01	RH	1	D	SE	1			5	1		P	51.47685	1.122096
16/12/2009	Transect 7	10:01	RH	5	A	E	1			10			S	51.47532	1.118965
16/12/2009	Transect 7	10:07	RH	1	D			1				flushed	P	51.46822	1.133986
16/12/2009	Transect 7	10:07	RH	1	C			1				flushed	S	51.466	1.129433
16/12/2009	Transect 7	10:08	RH	1	C			1				flushed	P	51.46617	1.134727
16/12/2009	Transect 7	10:09	RH	1	E	N	1			5			P	51.46661	1.140562
16/12/2009	Transect 7	10:12	RH	2	D	S	1			2[1]			P	51.4609	1.142916
16/12/2009	Transect 7	10:12	RH	1	D	NW	1			1[1]			S	51.458	1.137384
16/12/2009	Transect 7	10:21	RH	1	A	E	1			10[1]			S	51.44646	1.157239
16/12/2009	Transect 7	10:23	RH	1	E	NE	1			5			P	51.44658	1.166861
16/12/2009	Transect 8	09:43	RH	1	E	S	1			5			P	51.48064	1.121622
16/12/2009	Transect 8	09:43	RH	1	A	E	1			15			P	51.48309	1.127244
16/12/2009	Transect 8	09:30	RH	1	D	E	1			15			S	51.45893	1.161848
16/12/2009	Transect C1	09:16	BV	1	B			1				flushed	S	51.43222	1.194243
16/12/2009	Transect C2	08:59	RH	1	E	NW	1			2			P	51.44079	1.206188
16/12/2009	Transect C2	09:00	RH	1	D	SE	1			20			S	51.43488	1.200055
16/12/2009	Transect C2	09:01	RH	1	D	S	1			20			S	51.43332	1.202156
16/12/2009	Transect C2	09:06	BV	1	D			1		[1]		flushed	P	51.42847	1.218364
16/12/2009	Transect C3	08:49	RH	1	C			1				flushed	S	51.44885	1.206075
16/12/2009	Transect C4	08:28	RH	1	C			1				flushed	S	51.43799	1.231521
16/12/2009	Transect C4	08:31	RH	1	C			1				flushed	P	51.43508	1.241414
16/12/2009	Transect C4	08:31	RH	1	A	E	1			5[1]			S	51.43401	1.239566
16/12/2009	Transect C4	08:33	RH	1	C			1		[1]		flushed	S	51.43039	1.242481
11/01/2010	Transect 0	09:23	RH	1	A	W	1			2[1]			P	51.45717	1.016932
11/01/2010	Transect 0	09:27	RH	1	D			1		0[1]	1	Flushed.	S	51.44855	1.022457
11/01/2010	Transect 0	09:30	RH	1	B	SE	1			30	1		P	51.44558	1.033015
11/01/2010	Transect 0	09:31	BV	1	C			1		0[1]	1	Flushed.	S	51.44261	1.032961
11/01/2010	Transect 1	09:51	BV	1	D	W	1			30			S	51.42651	1.080414
11/01/2010	Transect 1	10:05	RH	1	B	W	1			3			S	51.44874	1.048593
11/01/2010	Transect 1	10:08	RH	1	C	SW	1			1			S	51.45404	1.043105
11/01/2010	Transect 1	10:10	RH	1	C	W	1			5			S	51.45702	1.03884
11/01/2010	Transect 1	10:10	RH	1	D			1		0	1		S	51.45756	1.039994
11/01/2010	Transect 1	10:12	RH	1	B	SE	1			10			S	51.45982	1.034017

Date	Transect	Time	Species	Flock	Distance Band	Direction	Fly	On Sea	Feed	Height	In Transect	Notes	Side	Latitude	Longitude
11/01/2010	Transect 1	10:12	RH	5	A	W	1			10			S	51.45956	1.033426
11/01/2010	Transect 1	10:12	RH	3	C			1		0	1		S	51.4602	1.034905
11/01/2010	Transect 1	10:12	RH	3	D			1		0	1	Flushed.	P	51.45814	1.030172
11/01/2010	Transect 1	10:13	RH	1	B			1		0	1		S	51.45982	1.034017
11/01/2010	Transect 1	10:13	RH	5	C	W	1			10			S	51.4602	1.034905
11/01/2010	Transect 1	10:13	RH	1	A			1		0	1		S	51.46126	1.031525
11/01/2010	Transect 1	10:14	RH	1	A			1		0	1		S	51.46289	1.029463
11/01/2010	Transect 1	10:14	RH	1	C	NE	1			10			S	51.46363	1.030825
11/01/2010	Transect 1	10:16	RH	1	D			1		0	1		S	51.46734	1.027747
11/01/2010	Transect 1	10:16	RH	1	A	E	1			10			S	51.46606	1.025221
11/01/2010	Transect 1	10:17	RH	1	B			1		0	1		S	51.46634	1.025782
11/01/2010	Transect 1	10:17	RH	1	D			1		0	1	Flushed.	S	51.46734	1.027747
11/01/2010	Transect 2	10:35	RH	1	A			1		0	1		S	51.46799	1.038622
11/01/2010	Transect 2	10:39	RH	1	B			1		0	1		P	51.46312	1.046234
11/01/2010	Transect 2	10:39	RH	1	A			1		0	1	Flushed.	S	51.4608	1.047313
11/01/2010	Transect 2	10:39	RH	1	D			1		0	1	Flushed.	P	51.46239	1.050356
11/01/2010	Transect 2	10:41	BV	1	A	NW	1			5			S	51.45729	1.052003
11/01/2010	Transect 2	10:42	RH	1	D	NW	1			5			S	51.45422	1.051884
11/01/2010	Transect 2	10:42	RH	1	C			1		0	1		S	51.4548	1.052991
11/01/2010	Transect 2	10:42	RH	11	A	W	1			10			S	51.45553	1.054373
11/01/2010	Transect 2	10:42	RH	1	D	W	1			3	1		P	51.45712	1.057416
11/01/2010	Transect 2	10:44	RH	2	C	SW	1			3			P	51.45475	1.058684
11/01/2010	Transect 2	10:44	RH	18	A	W	1			10			S	51.45201	1.059033
11/01/2010	Transect 2	10:44	RH	2	D	S	1			5			P	51.4536	1.062076
11/01/2010	Transect 2	10:45	RH	7	A	W	1			10			S	51.45027	1.061348
11/01/2010	Transect 2	10:52	RH	3	D	W	1			25			S	51.43689	1.075626
11/01/2010	Transect 2	10:54	RH	1	A	W	1			15	1		P	51.43493	1.0835
11/01/2010	Transect 2	10:58	RH	2	C	N	1			25			P	51.42858	1.094349
11/01/2010	Transect 3	11:27	RH	1	D			1		0	1	On sea between turbines KF20 & KF26 then flushed then low W out of turbine array.	S	51.46098	1.073018
11/01/2010	Transect 3	11:27	RH	1	C			1		0	1	Flushed.	P	51.45892	1.068285
11/01/2010	Transect 3	11:35	RH	1	A			1		0	1		S	51.47196	1.052964
11/01/2010	Transect 3	11:35	RH	4	A	W	1			5			S	51.47196	1.052964
11/01/2010	Transect 3	11:35	RH	2	C			1		0	1		S	51.47273	1.054285
11/01/2010	Transect 3	11:39	RH	2	E	SW	1			2			P	51.47533	1.038522
11/01/2010	Transect 3	11:41	RH	1	C	NE	1			5			P	51.47999	1.039368
11/01/2010	Transect 4	11:53	RH	2	D			1		0	1	Flushed.	P	51.47858	1.066686
11/01/2010	Transect 4	11:53	RH	1	A	W	1			6			P	51.47725	1.064233
11/01/2010	Transect 4	11:56	RH	1	A	SW	1			20			S	51.47176	1.070797
11/01/2010	Transect 5	12:35	RH	1	A			1		0	1		S	51.44413	1.124904

Date	Transect	Time	Species	Flock	Distance Band	Direction	Fly	On Sea	Feed	Height	In Transect	Notes	Side	Latitude	Longitude
11/01/2010	Transect 5	12:56	RH	2	D	W	1			10			S	51.47829	1.083847
11/01/2010	Transect 5	12:57	RH	1	D	N	1			10			S	51.47987	1.081757
11/01/2010	Transect 5	12:57	RH	2	D	W	1			10			S	51.47987	1.081757
11/01/2010	Transect 5	12:57	RH	4	A	W	1			10			S	51.47859	1.079231
11/01/2010	Transect 5	13:00	RH	2	C			1		0	1		P	51.48239	1.071249
11/01/2010	Transect 5	13:01	RH	1	B			1		0	1		S	51.48524	1.071476
11/01/2010	Transect 7	14:11	RH	1	A	W	1			3			P	51.48104	1.111583
11/01/2010	Transect 8	14:26	div	1	E			1		0			P	51.47286	1.148208
11/01/2010	Transect 8	14:32	RH	1	C			1		0	1	Flushed.	S	51.46155	1.152216
11/01/2010	Transect 8	14:34	RH	9	E			1		0		Flushed.	P	51.46356	1.161663
11/01/2010	Transect 8	14:34	RH	3	D	W	1			3			P	51.46046	1.161163
11/01/2010	Transect 8	14:35	RH	1	C			1		0	1	Flushed.	S	51.45638	1.159245
11/01/2010	Transect 8	14:39	RH	4	E	N	1			2			P	51.45317	1.17566
11/01/2010	Transect C1	14:44	RH	1	A	W	1			10			S	51.44149	1.180702
11/01/2010	Transect C1	14:46	RH	1	A	NE	1			5			S	51.43799	1.18502
11/01/2010	Transect C1	14:48	RH	1	D	W	1			30			P	51.43615	1.192425
11/01/2010	Transect C1	14:52	RH	5	D	W	1			20			S	51.42658	1.196171
11/01/2010	Transect C3	15:20	RH	1	C	N	1			5			P	51.44505	1.210561
11/01/2010	Transect C3	15:23	BV	1	D	W	1			5		Appeared to have just flushed.	S	51.43776	1.21377
11/01/2010	Transect C3	15:30	RH	1	D	W	1			1			S	51.42807	1.227967
11/01/2010	Transect C4	15:35	RH	1	D	N	1			1			S	51.43169	1.247554
11/01/2010	Transect C4	15:37	RH	3	A			1		0	1	Flushed.	S	51.43392	1.240658
11/01/2010	Transect C4	15:39	RH	1	E	N	1			2			P	51.43433	1.230373
11/01/2010	Transect C4	15:40	RH	7	E			1		0			P	51.4359	1.228211
11/01/2010	Transect C4	15:41	RH	1	C	W	1			1			S	51.44152	1.232624
19/01/2010	Transect 0	15:59	RH	1	A			1		0	1		S	51.45688	1.016742
19/01/2010	Transect 0	16:00	RH	3	A	W	1			2			S	51.45507	1.018939
19/01/2010	Transect 1	15:40	RH	1	D	W	1			5			P	51.46481	1.021554
19/01/2010	Transect 1	15:41	RH	1	A	SW	1			1			S	51.46798	1.022493
19/01/2010	Transect 4	13:41	RH	1	C			1		0	1		S	51.43497	1.118061
19/01/2010	Transect 4	13:44	RH	1	C	E	1			1			S	51.43097	1.123844
19/01/2010	Transect 5	12:35	RH	1	A	E	1			1			S	51.43427	1.137551
19/01/2010	Transect 6	12:15	RH	1	D	E	1			1	1		S	51.44757	1.131687
19/01/2010	Transect 6	12:16	RH	2	A	NE	1			12			S	51.44765	1.135986
19/01/2010	Transect 6	12:20	RH	1	A	NE	1			1			S	51.44211	1.143472
19/01/2010	Transect 6	12:24	RH	1	A	NE	1			10			S	51.43688	1.151084
19/01/2010	Transect 8	09:40	RH	1	D	N	1			2			P	51.46539	1.151933
03/02/2010	Transect 0	15:38	RH	2	C			1		0	1	Flushed.	S	51.46302	1.00694
03/02/2010	Transect 0	15:39	RH	3	C			1		0	1	Flushed.	S	51.46133	1.008987
03/02/2010	Transect 0	15:41	RH	1	A			1		0	1	Flushed.	S	51.45868	1.014549
03/02/2010	Transect 2	14:25	RH	1	A	E	1			25			P	51.46565	1.043038
03/02/2010	Transect 5	12:34	RH	2	C			1		0	1	Between turbines KF23 and	S	51.46303	1.1024

Date	Transect	Time	Species	Flock	Distance Band	Direction	Fly	On Sea	Feed	Height	In Transect	Notes	Side	Latitude	Longitude
												KF17.			
03/02/2010	Transect 5	12:45	RH	1	D			1		0	1	Flushed.	S	51.48453	1.075704
03/02/2010	Transect 6	11:54	RH	2	D			1		0	1		P	51.45932	1.125055
03/02/2010	Transect 6	12:00	RH	2	A	E	1			10	1		S	51.44927	1.133482
03/02/2010	Transect 6	12:07	RH	1	A			1		0	1	Flushed.	S	51.44097	1.144478
03/02/2010	Transect 6	12:12	RH	1	A	E	1			1			P	51.43313	1.155918
03/02/2010	Transect 7	11:13	RH	1	D	S	1			5			P	51.46045	1.134059
03/02/2010	Transect 7	11:14	RH	1	A			1		0	1	Flushed.	S	51.46381	1.135011
03/02/2010	Transect 8	09:20	RH	2	A	W	1			10			S	51.48055	1.130323
03/02/2010	Transect 8	09:20	RH	1	D	NE	1			8			P	51.48214	1.133367
03/02/2010	Transect 8	09:20	RH	1	A	N	1			15			S	51.48055	1.130323
03/02/2010	Transect 8	09:20	RH	1	E	NE	1			8			P	51.48358	1.136135
03/02/2010	Transect 8	09:22	div	6	E	W	1			10			P	51.48	1.141025
03/02/2010	Transect 8	09:23	RH	2	A	E	1			15			P	51.47725	1.135767
03/02/2010	Transect 8	09:23	RH	6	A	W	1			10	1		P	51.47539	1.138212
03/02/2010	Transect 8	09:24	RH	7	A	S	1			10			S	51.47332	1.138903
03/02/2010	Transect 8	09:24	RH	3	C			1		0	1	Flushed.	P	51.47307	1.1414
03/02/2010	Transect 8	09:26	RH	1	D	N	1			1			S	51.46874	1.134323
03/02/2010	Transect 8	09:30	RH	1	A			1		0	1		S	51.4627	1.147371
03/02/2010	Transect 8	09:31	RH	1	A			1		0	1		P	51.46227	1.151125
03/02/2010	Transect 8	09:35	RH	1	D	N	1			25	1		P	51.45767	1.164866
03/02/2010	Transect 8	09:36	RH	6	C			1		0	1		S	51.45366	1.162637
03/02/2010	Transect 8	09:36	RH	6	A	W	1			5			S	51.45437	1.164039
03/02/2010	Transect 8	09:37	RH	9	C			1		0	1	Flushed.	P	51.45361	1.168267
03/02/2010	Transect 8	09:38	RH	3	A	W	1			5			S	51.4509	1.168466
03/02/2010	Transect C1	09:46	RH	4	B			1		0	1	Flushed.	S	51.43675	1.185533
03/02/2010	Transect C1	09:51	RH	3	C			1		0	1		P	51.42963	1.199163
03/02/2010	Transect C1	09:52	RH	1	C	E	1			10			P	51.42794	1.201339
03/02/2010	Transect C1	09:53	RH	1	D	S	1			10			P	51.42681	1.204714
03/02/2010	Transect C1	09:55	RH	1	E			1		0			P	51.42767	1.207468
03/02/2010	Transect C2	10:06	RH	5	D			1		0	1	Flushed.	S	51.43703	1.207005
03/02/2010	Transect C4	10:42	RH	12	D			1		0	1		P	51.4397	1.227263
03/02/2010	Transect C4	10:42	RH	4	A			1		0	1	Flushed.	P	51.44263	1.227918
03/02/2010	Transect C4	10:45	RH	17	E			1		0		Flushed.	P	51.44574	1.215929
19/02/2010	Transect 0	09:30	RH	12	D	NE	1			15		2km ahead	P	51.468	1.008484
19/02/2010	Transect 0	09:32	RH	1	C	N	1			12			S	51.46234	1.007583
19/02/2010	Transect 0	09:37	RH	3	C	NE	1			5			S	51.45406	1.018725
19/02/2010	Transect 0	09:38	RH	1	A	W	1			2		1km ahead	P	51.45345	1.022912
19/02/2010	Transect 0	09:40	RH	1	D	SE	1			10		break in survey while deviating from line	S	51.44914	1.026947
19/02/2010	Transect 1	10:46	RH	1	A	W	1			1			S	51.4604	1.032908
19/02/2010	Transect 2	11:07	RH	1	C	NE	1			10	1		S	51.47137	1.030956
19/02/2010	Transect 2	11:37	RH	1	C			1		0	1	flushed	P	51.42557	1.097391

Date	Transect	Time	Species	Flock	Distance Band	Direction	Fly	On Sea	Feed	Height	In Transect	Notes	Side	Latitude	Longitude
19/02/2010	Transect 2	11:37	RH	1	A			1		0	1		S	51.42411	1.096445
19/02/2010	Transect 4	12:58	RH	1	D	N	1			1			S	51.43719	1.112766
19/02/2010	Transect 4	13:02	RH	2	B			1				flushed	P	51.4331	1.123606
19/02/2010	Transect 4	13:03	RH	1	D			1		0	1	flushed	S	51.43138	1.119811
19/02/2010	Transect 4	13:05	RH	1	E	E	1			5			P	51.43139	1.133707

1

Note: RH = red-throated diver, div = diver species, BV = black-throated diver.

Figure 1. Boat survey area used in 2009-10, showing the main transects and the additional transect extensions

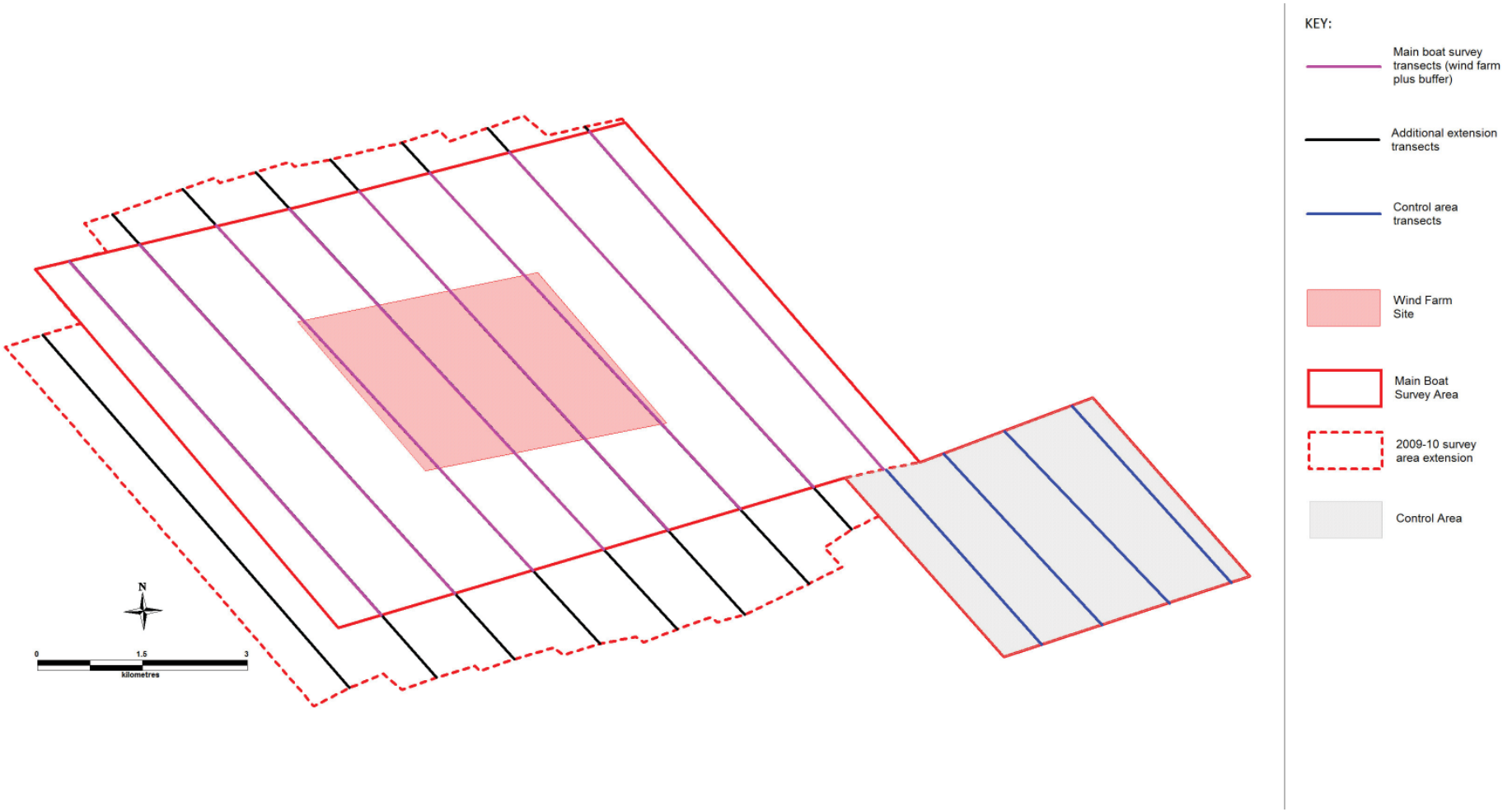


Figure 2. Diver population estimates for each survey of the wind farm plus buffer, control and new extension zones in 2009-10.

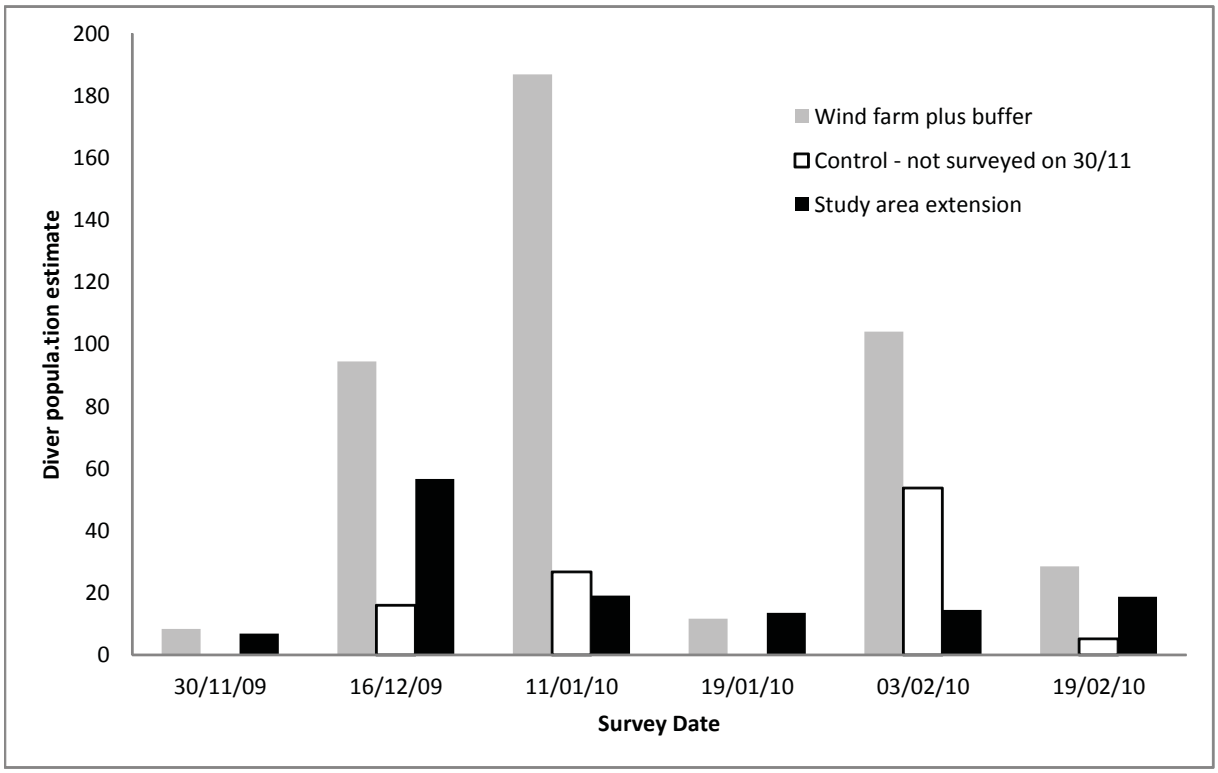


Figure 3a. Distribution of divers recorded during the 2009-10 post-construction boat surveys

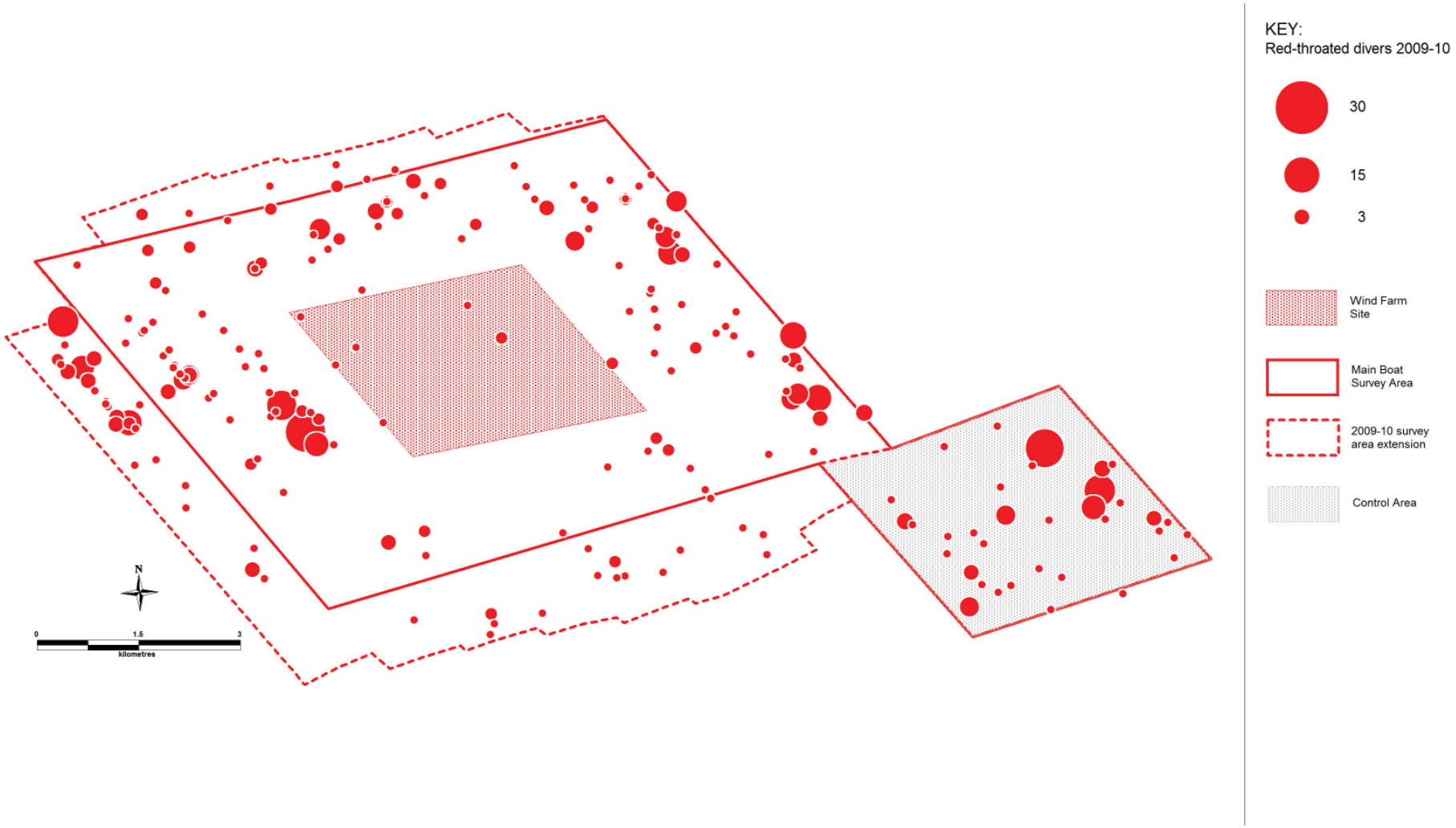


Figure 3b. Distribution of divers recorded during the pre-construction boat surveys

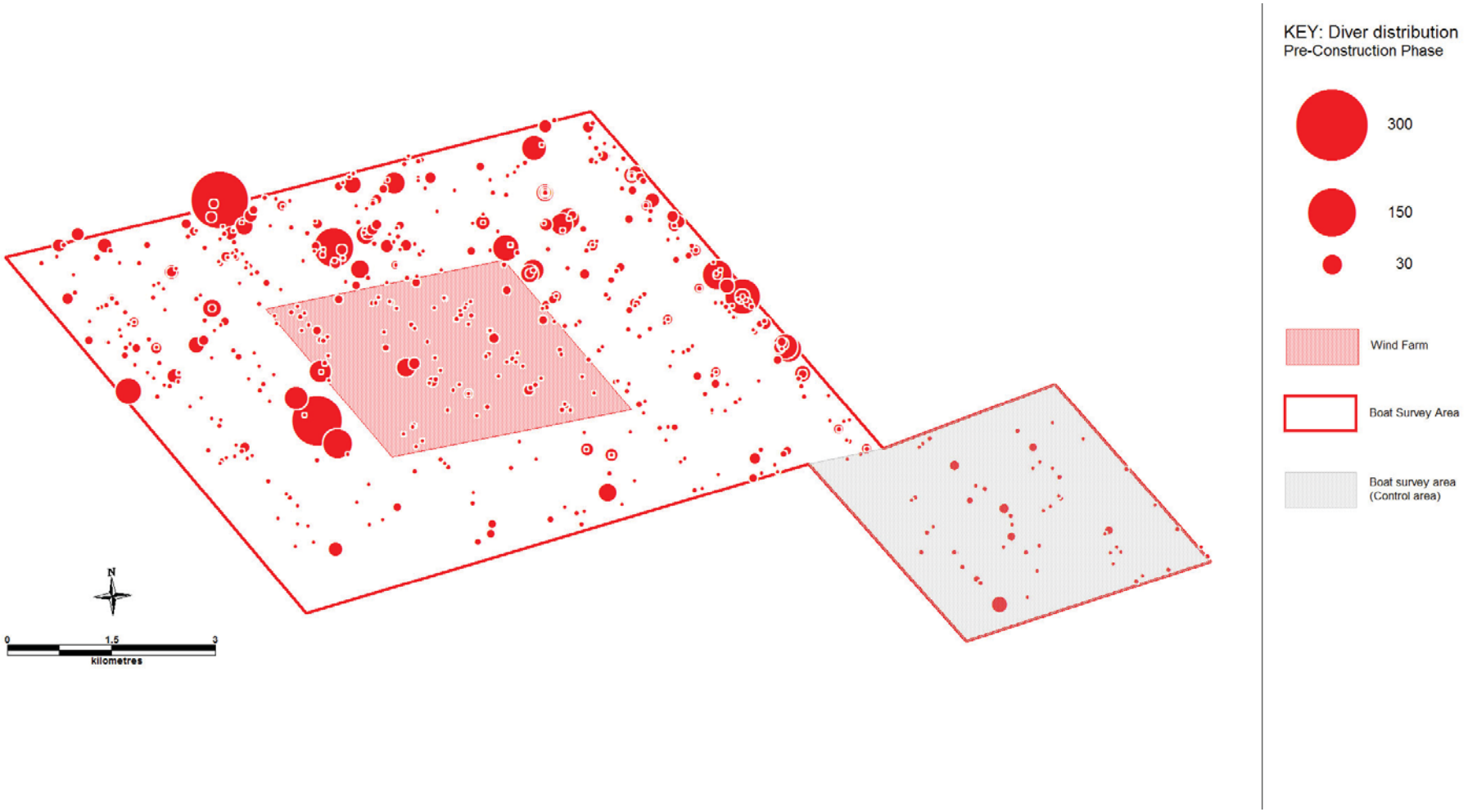


Figure 3c. Distribution of divers recorded during the construction phase boat surveys

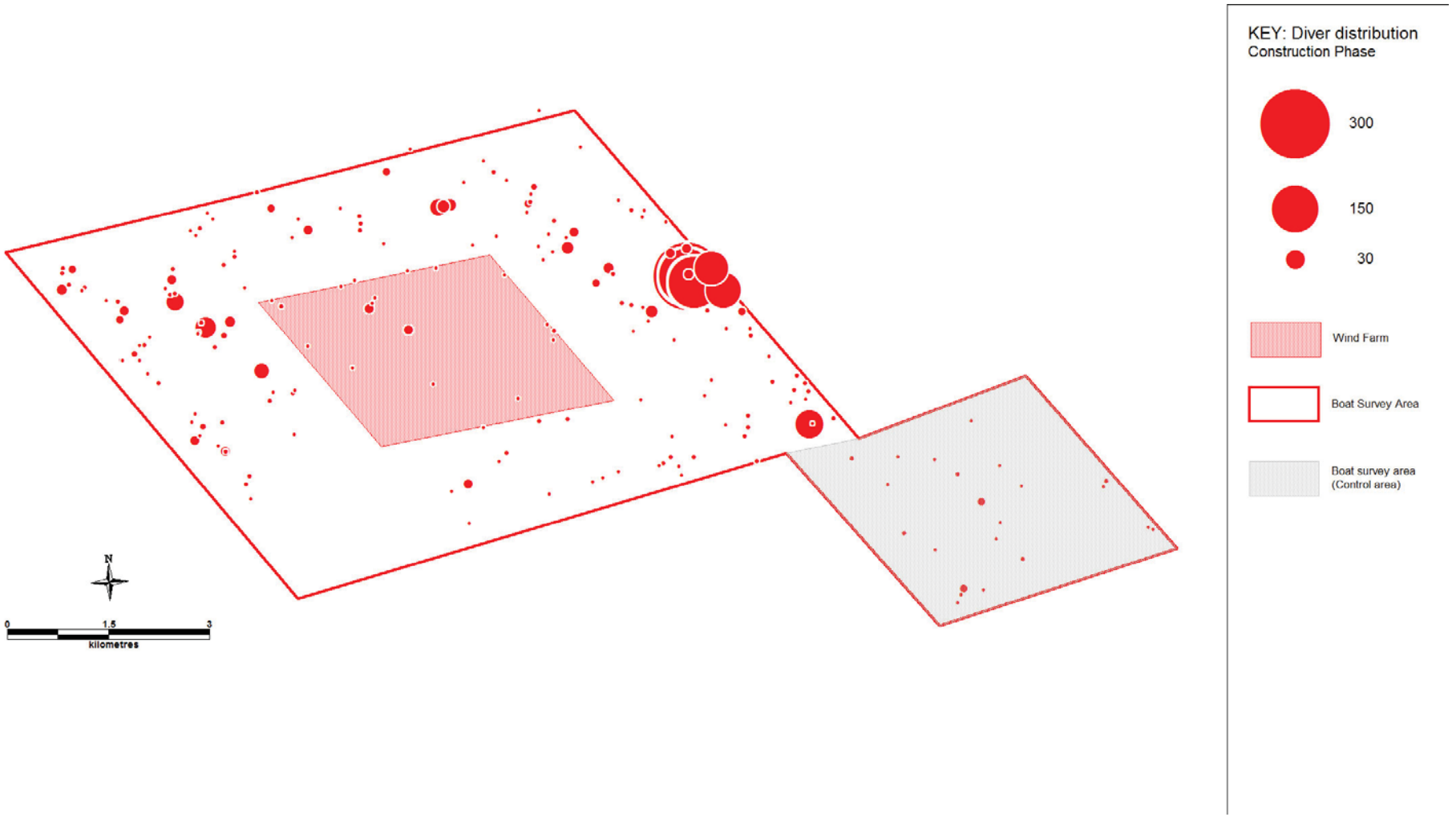


Figure 3d. Distribution of divers recorded during the post-construction boat surveys

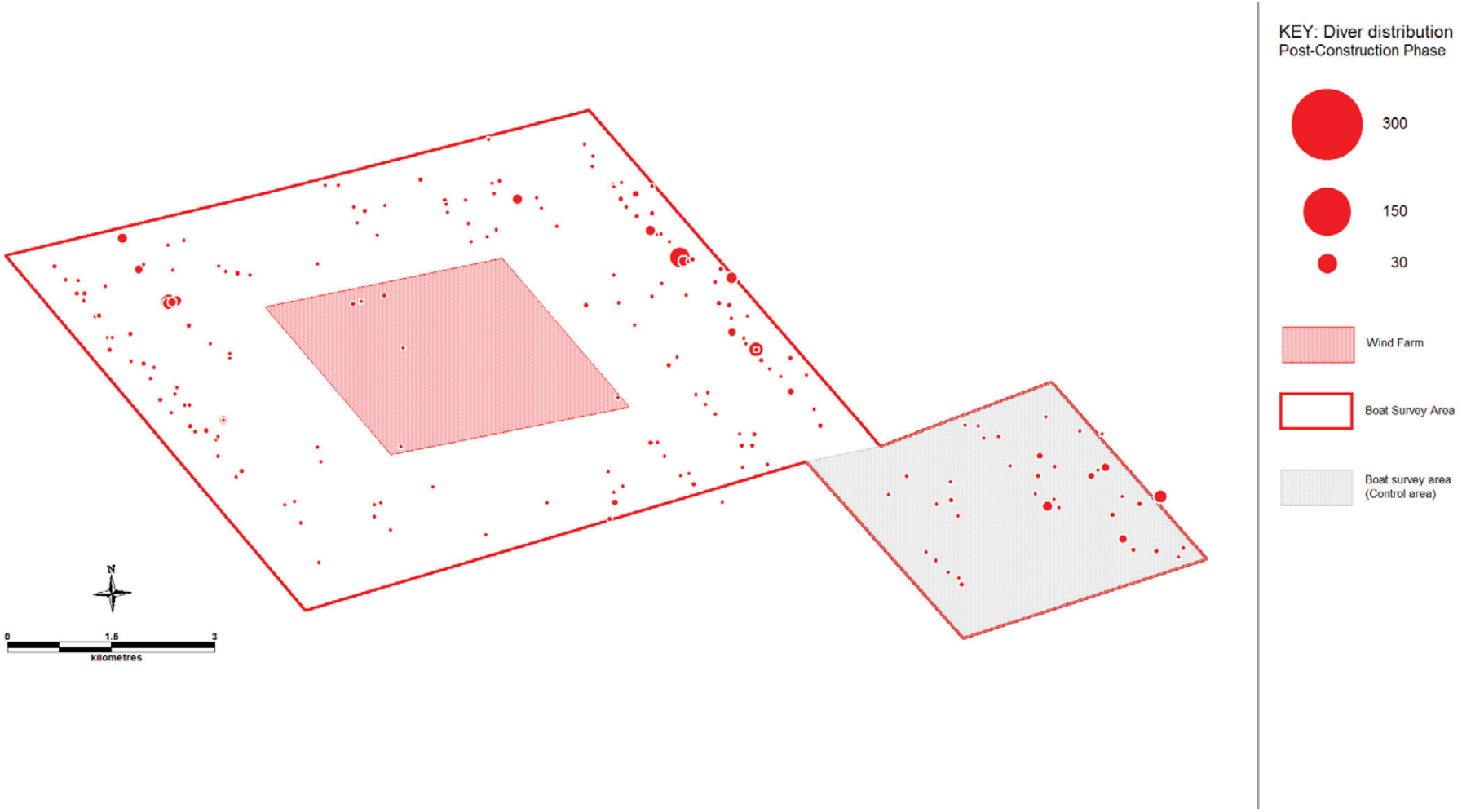


Figure 3e. Distribution of divers recorded during the 2008-09 post-construction boat surveys

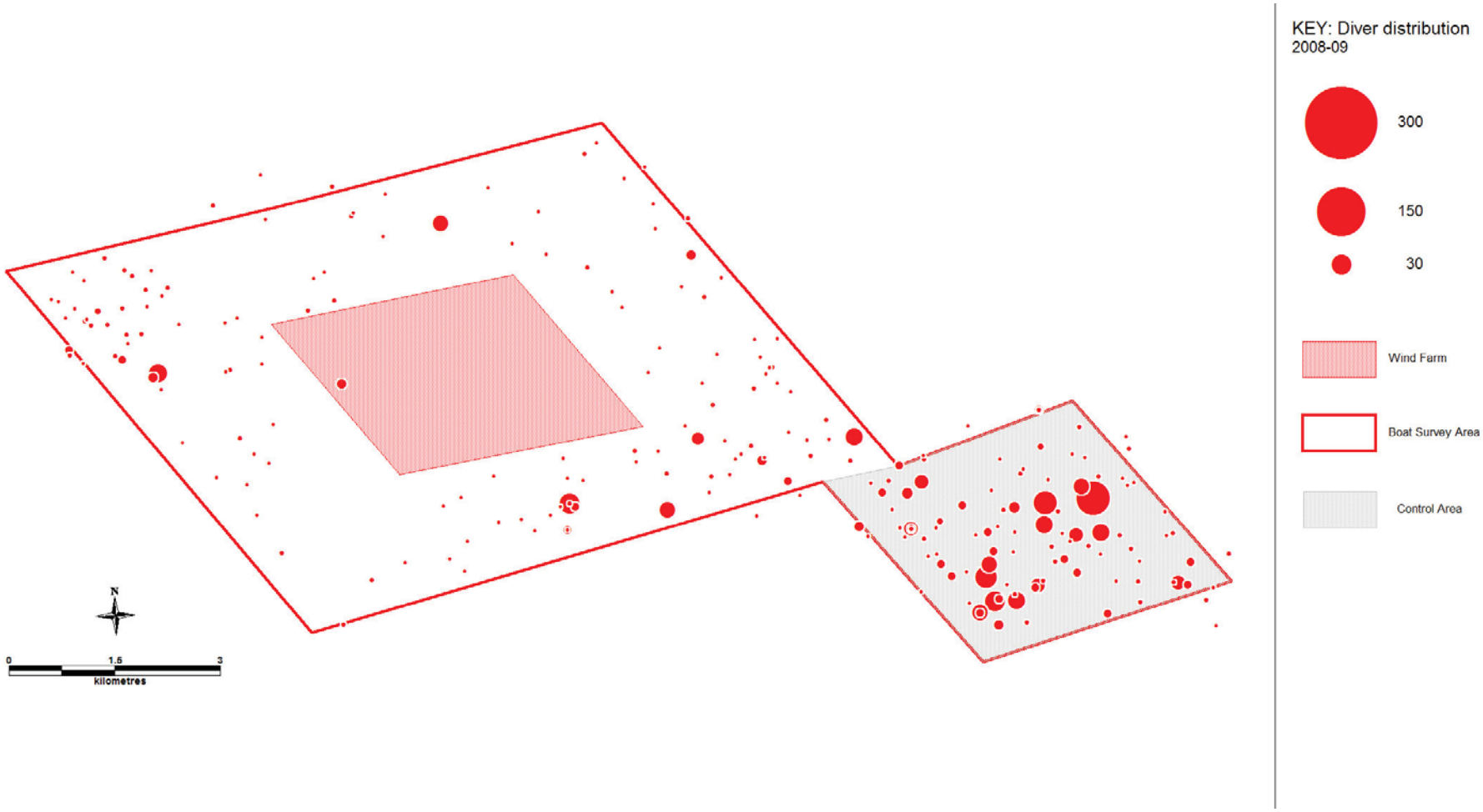


Figure 4. Diver population estimates for the wind farm footprint and surrounding buffer zones for each survey year (grey = pre-construction - 2002-03 and 2003-04, white = construction year - 2004-05 and black = post-construction - 2005-06, 2006-07, 2008-09 and 2009-10)

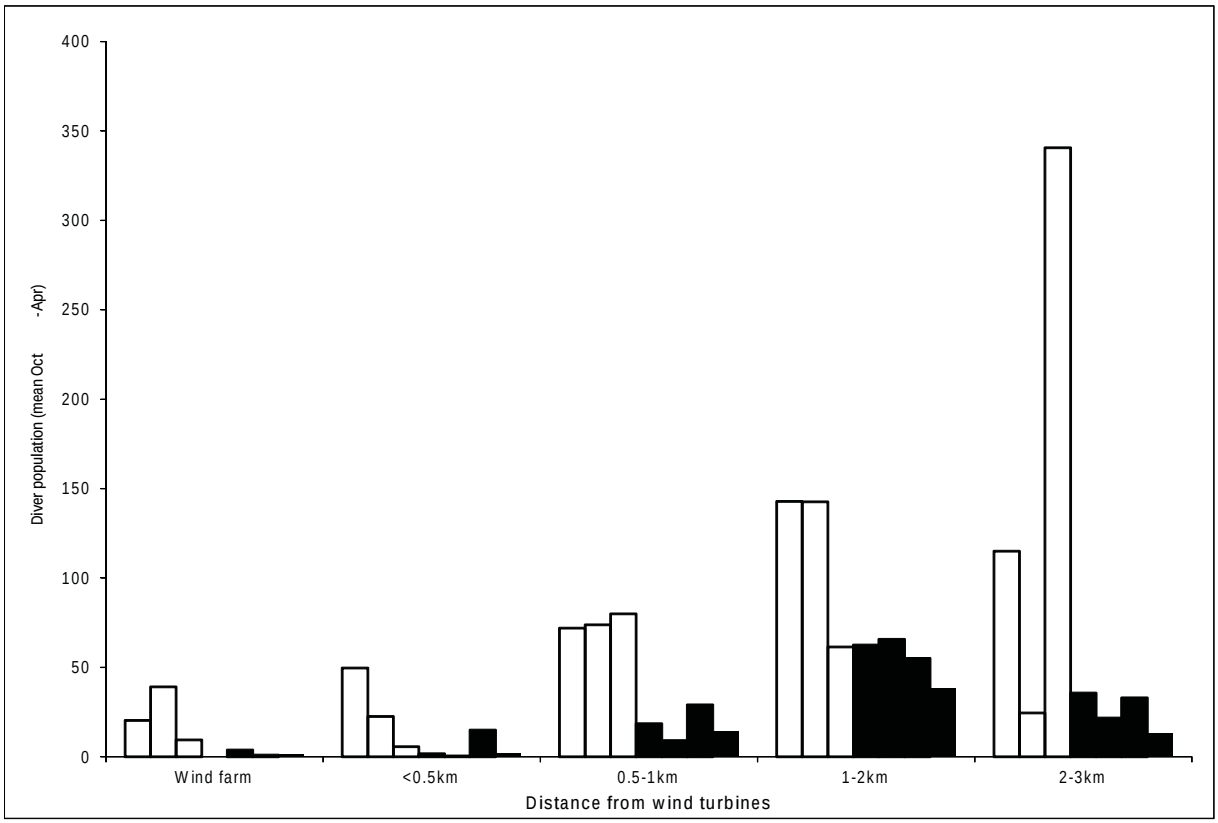


Figure 5 Diver population densities for the wind farm footprint and surrounding buffer zones for each survey year (grey = pre-construction - 2002-03 and 2003-04, white = construction year - 2004-05 and black = post-construction - 2005-06, 2006-07, 2008-09 and 2009-10)

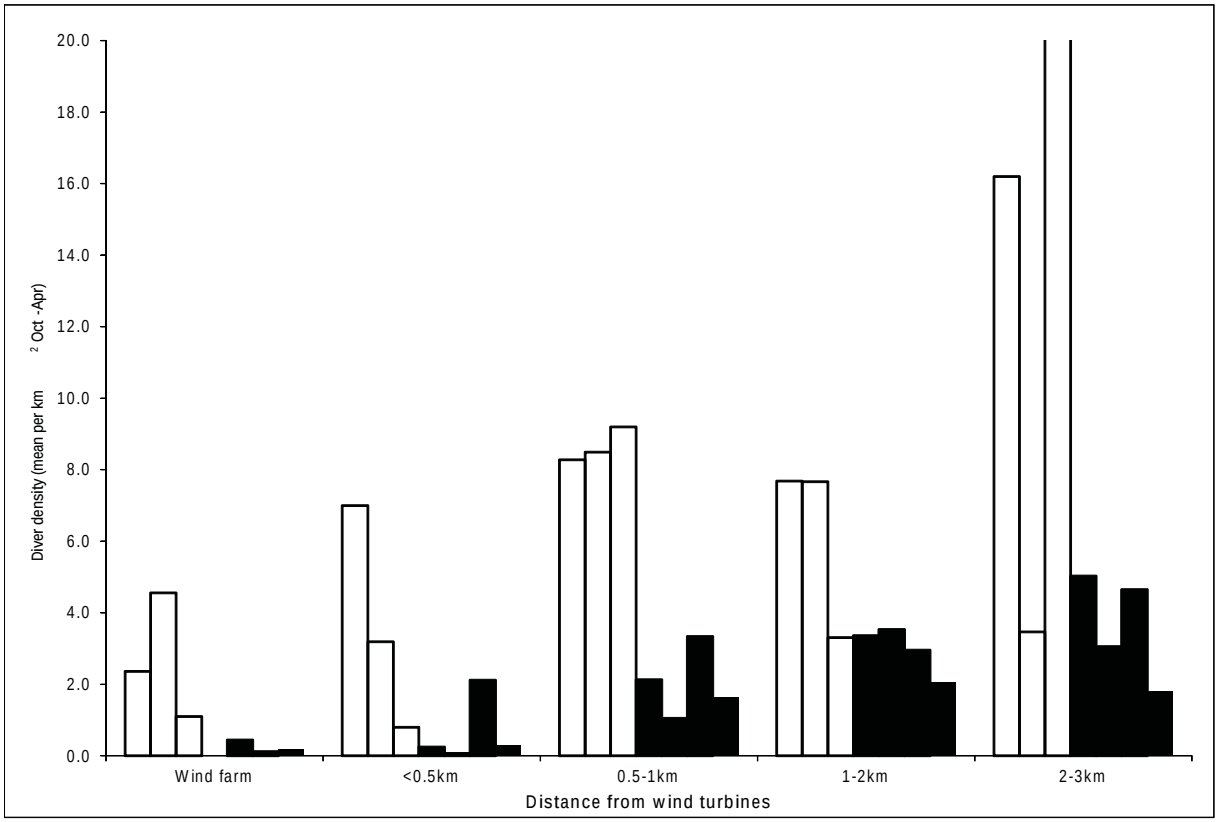


Figure 6. Diver proportionate distribution for the wind farm footprint and surrounding buffer zones for each survey year (grey = pre-construction - 2002-03 and 2003-04, white = construction year - 2004-05 and black = post-construction - 2005-06, 2006-07, 20008-09 and 2009-10). Red dashed lines indicate expected proportion if distribution uniform.

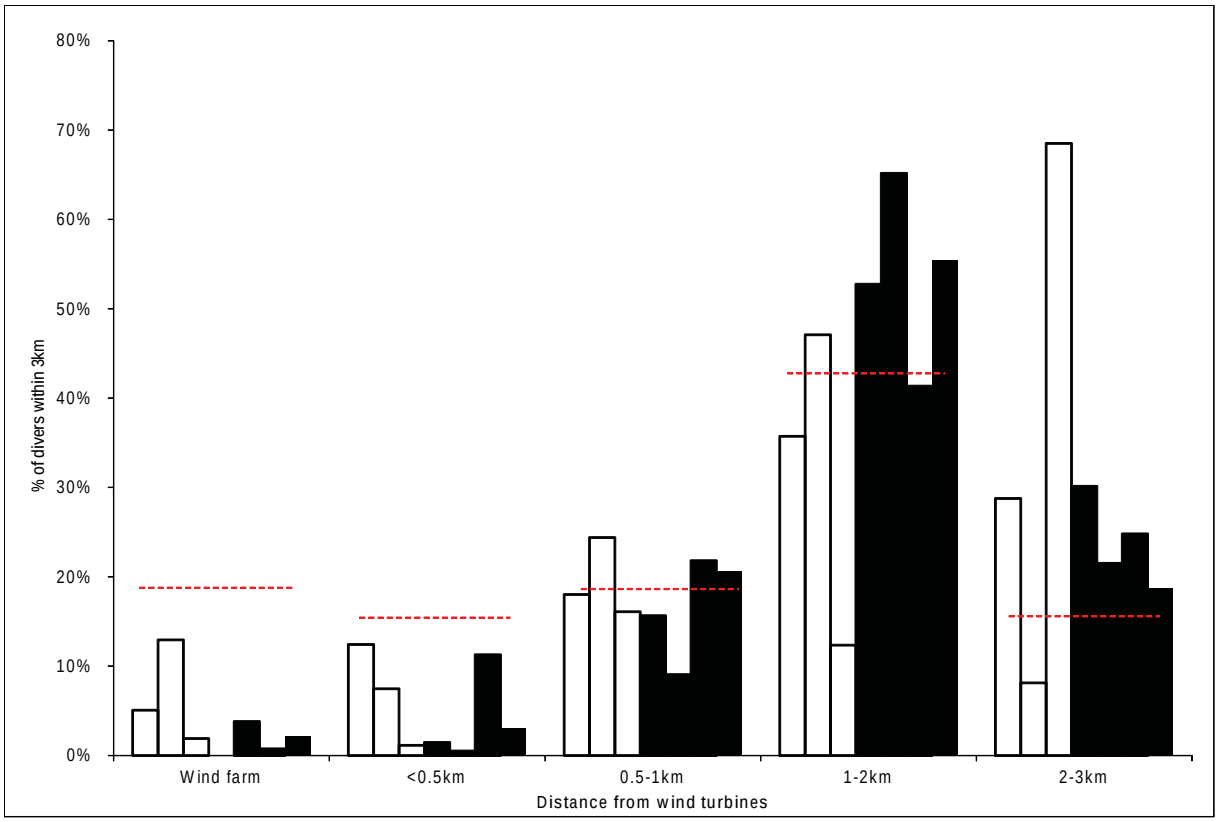
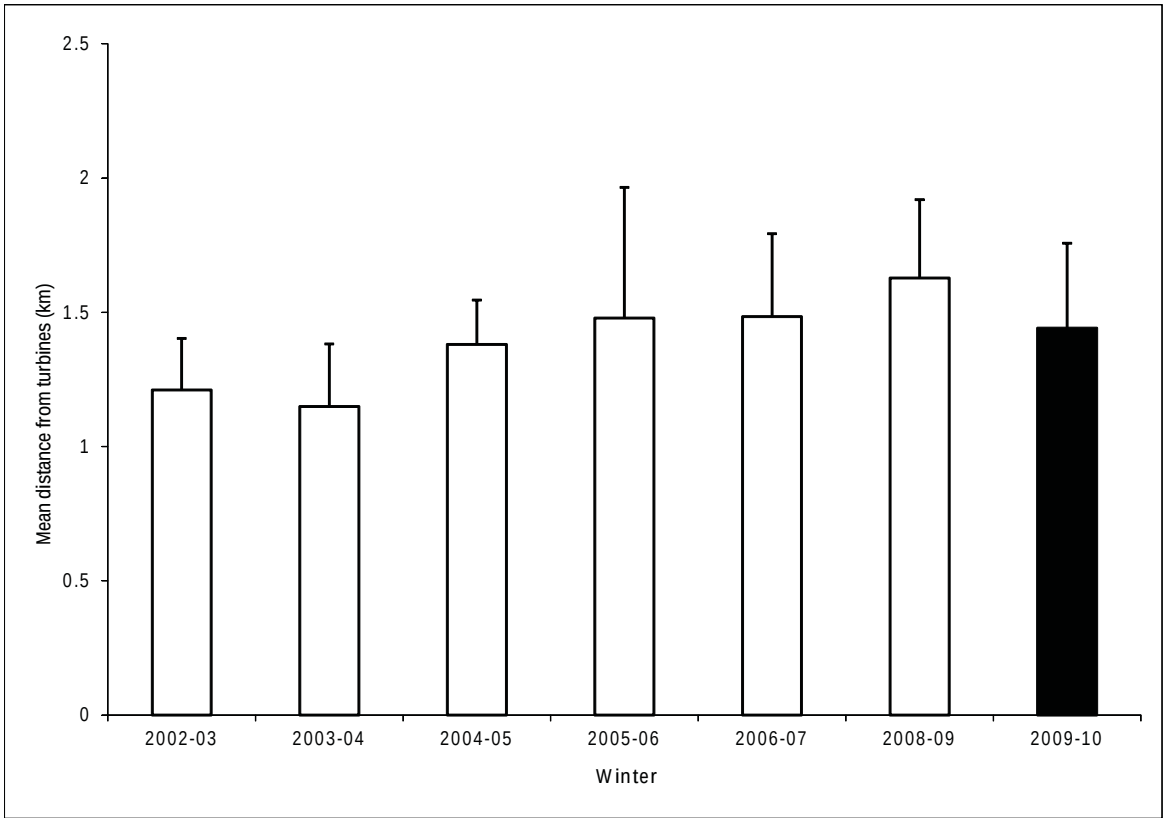


Figure 7. Distances between diver records and the wind turbine locations during each winter of the pre-construction (grey bars), construction (white) and post-construction (black) phases. Mean values are shown + upper 95% confidence limit



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Effects of the construction of Scroby Sands offshore wind farm on the prey base of Little tern *Sternula albifrons* at its most important UK colony

Martin R. Perrow, James J. Gilroy [1], Eleanor R. Skeate, Mark L. Tomlinson

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Abstract

Despite widespread interest in the impacts of wind farms upon birds, few researchers have examined the potential for indirect or trophic (predator–prey) effects. Using surface trawls, we monitored prey abundance before and after construction of a 30 turbine offshore wind farm sited close to an internationally important colony of Little terns. Observations confirmed that young-of-the-year clupeids dominated chick diet, which trawl samples suggested were mainly herring. Multivariate modelling indicated a significant reduction in herring abundance from 2004 onwards that could not be explained by environmental factors. Intensely noisy monopile installation during the winter spawning period was suggested to be responsible. Reduced prey abundance corresponded with a significant decline in Little tern foraging success. Unprecedented egg abandonment and lack of chick hatching tentatively suggested a colony-scale response in some years. We urge a precautionary approach to the timing and duration of pile-driving activity supported with long-term targeted monitoring of sensitive receptors.

Highlights

▸ First evidence of indirect effect of wind farm construction on a seabird via its prey. ▸ Noise generated by pile driving thought responsible for the decline in young herring. ▸ Concomitant significant reduction in foraging success of Little terns. ▸ Circumstantial evidence of population response with unprecedented egg abandonment. ▸ Call for caution in relation to pile-driving and longer-term monitoring of effects.

Previous Next

FEEDBACK

Keywords

Herring *Clupea harengus*; Pile driving; Indirect effects; Trophic interactions; Impact assessment

Recommended articles Citing articles (21)

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2. Methodology

2.1. Study site

2.2. Sampling the prey resource of Little terns

2.3. Foraging success of Little terns

2.4. Prey type in chick provisions

2.5. Statistical analysis

3. Results

3.1. Variation in prey abundance

3.2. Chick provisioning and foraging success

4. Discussion

4.1. Limitations of the monitoring programme

4.2. Factors affecting recruitment of clupeids

4.3. Potential impact of wind farm construction

5. Concluding remarks

Acknowledgements

References

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Ministry of the Environment · Denmark

Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark

NERI Report
Commissioned by DONG energy and Vattenfall A/S
2006



National Environmental Research Institute
Ministry of the Environment · Denmark

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2006

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National Environmental Research Institute

Summary

This report presents data on monitoring investigations of birds carried out during 1999-2005 in relation to the construction of the world's first two large offshore wind farms at Horns Rev and Nysted in Denmark. We consider the hazards turbines posed to birds and the physical and ecological effects that these cause. We propose a series of hypotheses relating to these effects on birds at the two sites, testing to see if birds do indeed show reactions to the turbines once erected, relative to their "unaffected" behaviour we monitored during pre-construction baseline studies. In this way, the effects of the construction of the wind farms at sea could be predicted from our hypotheses and validated by post construction monitoring and data collection which was a condition of planning permission for the Danish projects. Throughout, we have restricted our studies primarily to waterbirds, because these are the species that exploit the offshore environment in general and the two study areas in particular, because Denmark has a special responsibility for the maintenance of their populations and the habitat that they use and because long lived birds with relatively low annual breeding success (which include many waterbirds) are those most susceptible to additional mortality. This does not mean that other species (such as many bird of prey and short-lived land birds that pass through the areas on migration) are not important, but their study was generally beyond the scope of these investigations.

In general terms, the potential effects of the construction of a wind farm on birds were considered to arise from three major processes:

1. A behavioural element caused by birds avoiding the vicinity of the turbines as a behavioural response to a visual (or other) stimulus. This can have two effects:
   - a barrier effect affecting bird movement patterns, potentially increasing costs
   - the displacement of birds from favoured distribution, equivalent to habitat loss
2. Physical changes due to construction (physical habitat loss, modification to bottom flora and fauna and creation of novel habitats, e.g. for resting on the static superstructure).
3. A direct demographic element resulting from physical collision with the superstructure (mortality).

We investigated each of these elements at both wind farms collecting data prior to and post construction as outlined under the following headings.

1a) At Horns Rev and Nysted, the barrier effect was studied by mapped bird migration routes combining radar techniques by day and night with specific species identification during daylight hours using telescopes. Radar tracks were entered to a GIS platform to compare the base-line with subsequent post-construction monitoring results. Emphasis was placed upon three key variables:

I. the orientation of migration routes for waterbirds and terrestrial species to measure potential avoidance responses and response distances,
II. the probability that waterbirds will pass through the wind farm area to measure waterbird responses to the entire wind farm,
III. migration intensity, measured by the number of bird flocks that pass into the wind farm area, to measure the effect of avoidance responses on the volume of migration within the wind farm area post construction.

Comparisons of these key variables between individual base-line years were undertaken by controlling for various factors such as weather conditions, season and time of day using multi-factor analysis of variance (ANOVA) and regression analyses.

Results showed birds generally avoided Horns Rev and Nysted wind farms, although responses were highly species specific. Some species (e.g. divers and Gannets) were almost never seen flying between turbines, others rarely (e.g. Common Scoter) whilst others showed little avoidance behaviour (e.g. Cormorants and gulls). Overall, at Horns Rev, 71-86% of all bird flocks heading for the wind farm at 1.5-2 km distance avoided entering into the wind farm between the turbine rows patterns confirmed at Nysted (78%), predominantly amongst waterbirds. There was considerable movement of birds along the periphery of both wind farms, as birds preferentially flew around rather than between the turbines. Such avoidance was calculated to add an additional period of flight equivalent to an extra 0.5-0.7% on normal migration costs of Eiders migrating through Nysted. Changes in flight direction tended to occur closer to the wind farm by night than day at both sites, but avoidance rates remained high in darkness, when it was also shown birds tend to fly higher. Few data on avoidance behaviours were available during conditions of poor visibility, because intense migration generally slows and ceases during such conditions.

1b) Comparison of pre- and post construction aerial surveys of waterbird abundance and distribution in and around the two Danish offshore wind farms generally showed that they avoided the turbines (at least during the three years following construction), although responses were highly species specific. Divers at Horns Rev showed complete avoidance of the wind farm area during the three years post construction period, despite being present in average densities prior to construction. Common Scoter were absent from Horns Rev pre-construction, but occurred in large numbers in the vicinity of the wind farm, but were almost never seen within the turbines despite up to 381,000 in the general area. Terns and auks also occurred in the area but were almost never seen within

the Horns Rev wind farm post erection. Long-tailed Ducks showed statistically significant reductions in density post construction in the Nysted wind farm (and in sectors 2 km outside) where they had shown higher than average densities prior to construction. This strongly suggests major displacement of this species from formerly favoured feeding areas, although the absolute numbers were relatively small and therefore of no significance to the population overall. No bird species demonstrated enhanced use of the waters within the two Danish offshore wind farms after the erection of turbines, but it was clear, for example amongst Cormorants at Nysted, that the wind farm area was used occasionally for social feeding by very large numbers of birds post construction. Although bird displacement (as a result of behavioural avoidance of wind farms) represents effective habitat loss, it is important to assess the relative loss in terms of the proportion of potential feeding habitat (and hence the proportion of birds) affected relative to the areas outside of the wind farm. For most of the species considered here, the proportion is relatively small and therefore likely of little biological consequence. However, the additional cost arising from the construction of many other such wind farms may constitute a more significant effect. Hence, consideration of such cumulative effects of many such developments along an avian flyway represents an important priority in the future.

2) Physical habitat loss and gain was considered trivial, since even accounting for the anti-scour structures, the extent of the change equated to less that 1% of the total area of marine substrate enclosed within the total wind farm. Their effects would therefore be small and difficult to distinguish from other distributional effects described by monitoring changes in bird densities, except for the arrival of new species (which was not observed during these two studies) attracted to novel habitats post construction.

3) The avoidance responses documented above mean that although turbine construction at sea has a major effect on the local (i.e. wind farm project level) distribution, abundance and flight patterns of birds, the corollary is that many fewer birds come within the risk zone of the rotor blade sweep zone. Radar study results demonstrated that birds may show avoidance responses up to 5 km from the turbines, and that >50% of birds heading for the wind farm avoid passing within it. Radar studies at Horns Rev and Nysted also confirm that many birds entering the wind farm reorientate to fly down between turbine rows, frequently equidistant between turbines, further minimising collision risk. The Nysted Thermal Animal Detection System (TADS a remote infra red video monitoring system) and radar studies confirmed that waterbirds (mostly Eider) reduced their flight altitude within the wind farm, flying more often below rotor height than they did outside the wind farm. A stochastic predictive collision model was developed to estimate the numbers of Eiders, the most common species in the area, likely to collide with the sweeping turbine blades each autumn at the Nysted wind farm. Using parameters (including those described above) derived from radar investigations and TADS, and 1,000 iterations of the model, it was predicted with 95% certainty that out of 235,000 passing birds, 0.018-0.020% would collide with all turbines in a single autumn (41-48 individuals), equivalent to less than 0.05% of the annual hunt in Denmark (currently c. 70,000 birds). With such a low level of probability of collision expected at any one turbine, it was predicted that the TADS monitoring system would fail to detect a single collision of a waterbird during more than 2,400 hours of monitoring that was undertaken at the site, and this proved to be the case. This level of monitoring resulted in a mere 11 bird detections well away from the sweep area of the turbine blades, 2 passing bats, two passing objects that were either small birds or bats, a moth and one collision of a small bird.

The Planning Inspectorate

Advice Note 11, Annex C – Natural England and the Planning Inspectorate

Introduction

Part 1 of the Planning Inspectorate Advice Note 11: "Working with Public Bodies" covers many of the generic points of interaction relevant to the Planning Inspectorate (the Inspectorate) and Public Bodies. This Annex C to Advice Note 11 helps applicants understand Natural England's particular role in infrastructure planning under the Planning Act 2008 (PA2008). It explains:

- What is Natural England?
- What does Natural England do?
- Natural England's role in Nationally Significant Infrastructure Projects
    - Environmental opportunities
    - Natural England's engagement in the six NSIP stages
    - Environmental Impact Assessments (EIA)
    - Habitats Regulations Assessments (HRA)
    - Sites of Special Scientific Interest (SSSI)
    - Wildlife Licensing
    - Designated landscapes
    - Soils and best and most versatile agricultural land
- How to contact Natural England

This Annex will be kept under review and was updated in May 2018. The Planning Inspectorate and Natural England welcome feedback on the content of this Annex.

What is Natural England?

Natural England is a statutory nature conservation body (SNCB) established by the Natural Environment and Rural Communities Act 2006. (NERC Act) Natural England's general purpose is to ensure that the natural environment is conserved, enhanced, and managed for the benefit of present and future generations, thereby contributing to sustainable development. Natural England is financed by the Department for the Environment Food and Rural Affairs (Defra) but is a Non-Departmental Public Body, which forms its own views based on the best scientific evidence available.

What does Natural England do?

Natural England works for people, places and nature, to enhance biodiversity, landscapes and wildlife in rural, urban, coastal and marine areas; promoting access, recreation and public well-being, and contributing to the way natural resources are managed so that they can be enjoyed now and by future generations.

Section 2 of the NERC Act provides that Natural England's general statutory purpose is:
'… to ensure that the natural environment is conserved, enhanced and managed for the benefit of present and future generations, thereby contributing to sustainable development.'
This includes:

- promoting nature conservation and protecting biodiversity;
- conserving and enhancing the landscape;
- securing the provision and improvement of facilities for the study, understanding and enjoyment of the natural environment;
- promoting access to the countryside and open spaces and encouraging open-air recreation; and
- contributing in other ways, to social and economic well-being through management of the natural environment.

Natural England is required to keep under review all matters relating to its general purpose, and to provide public authorities with advice where they request this.

Natural England's role in Nationally Significant Infrastructure Project's (NSIPs)
Natural England's remit, responsibilities and geographical extent in relation to NSIPs comprises

- Advice on impacts on protected nature conservation sites[1] and designated landscapes[2] within England and out to 12 nautical miles of the English coastline.

- Its role as the consenting and licensing body for protected species licensing in the terrestrial environment in England (this applies to all activities undertaken landward of the mean low water mark). The Marine Management Organisation (MMO) is responsible for licensing seaward of the mean low water mark.

- There may also be cross border situations where, dependent on the nature and location of the proposal, Natural England will have a joint duty regarding protected sites with Natural Resources Wales (NRW) or Scottish Natural Heritage (SNH).

- The Joint Nature Conservation Committee (JNCC) has delegated its role as the SNCB for **offshore renewable energy projects** (wave, wind and tide) in English offshore waters, outside 12nm, to Natural England. This means that all provision of advice on renewable energy projects in inshore and offshore waters, adjacent to England (0-200nm), is provided by Natural England rather than being split between JNCC and Natural England at the 12nm boundary.

- For NSIPs that are not renewable energy projects beyond 12 nautical miles of the English coastline, Natural England will support JNCC in the exercise of its duties under the Habitats Regulations, particularly when there are impacts on protected sites inside 12 nautical miles.

- Advice as a statutory consultee for development of 'best and most versatile' agricultural land and for the reclamation of minerals and waste sites to agriculture.

- Advice on any other issues within Natural England's statutory remit e.g. on green infrastructure or environmental enhancement opportunities.

[1] Special Protection Areas, Special Areas of Conservation, Ramsar Sites, Sites of Special Scientific Interest, Marine Conservation Zones
[2] National Parks, The Broads and Areas of Outstanding Natural Beauty

Natural England's main responsibilities relate to providing advice on EIA, the Habitats Regulations, the regulation of Sites of Special Scientific Interest (SSSI) under the Wildlife and Countryside Act 1981 (as amended) (WCA), as the licensing body in respect of protected species, and as a consultee in relation to development of best and most versatile agricultural land or mineral and waste site reclamation to agriculture and as an adviser on landscape matters.

Environmental opportunities

Natural England will seek opportunities for positive environmental outcomes from major infrastructure developments. NSIPs can make a significant contribution to delivering the environmental ambition in the Government's 25 Year Environment Plan (25YEP). This aims to deliver an environmental net gain through development and infrastructure.

We can help applicants and the Examining Authority to better understand and value the benefits derived from the natural environment ('natural capital'). We may advise on opportunities to secure positive environmental benefits from NSIPs. Priorities include delivering high quality, multi-functional green infrastructure, establishing more coherent and resilient ecological networks and providing and enhancing habitats for protected species. We can also advise on approaches and metrics that enable projects to achieve biodiversity net gain, as set out in the National Planning Policy Framework and the recent and developing National Policy Statements, and on approaches to achieving wider natural capital gains.

Natural England will input into all phases of the PA2008 process as required, focusing mainly on pre-application, and examination. The main roles and responsibilities of Natural England under the 2008 Act fall into the following categories:

- as one of the prescribed consultees under section 42 of the 2008 Act that applicants are required to consult before submitting a NSIP application[3]

- as one of the consultation bodies that the Inspectorate must consult before adopting a scoping opinion in relation to any Environmental Impact Assessment (EIA)[4] and as a prescribed consultee for the environmental information submitted pursuant to the Infrastructure Planning (Environmental Impact Assessment) Regulations 2017.

- as a statutory party in the examination of Development Consent Order (DCO) applications[5]

- as a SNCB under the Conservation of Habitats and Species Regulations 2017 (Habitats Regulations) or the Conservation of Offshore Marine Habitats and Species Regulations 2017 (Offshore Regulations) in respect of Habitats Regulations Assessment (HRA) .

- as a consenting and licensing body/authority in respect of protected species and operations likely to damage the protected features of SSSIs pursuant to the Wildlife

[3] Under s. 42(a) Planning Act 2008 and Schedule 1 of the Infrastructure Planning (Applications: Prescribed Forms and Procedure) Regulations 2009 (as amended), at: http://infrastructure.planningportal.gov.uk/wp-content/uploads/2009/08/uksi_20092263_en.pdf .

[4] Under Regulation 10 of the Infrastructure Planning (Environmental Impact Assessment) Regulations 2017

[5] Under s. 88(3) (c) and s.102(ca) Planning Act 2008 and the Infrastructure Planning (Interested Parties and Miscellaneous Prescribed Provisions) Regulations 2015.

and Countryside Act 1981 (as amended) (WCA 1981)[6] and in relation to European protected species under the Habitats Regulations.

- As a prescribed consultee under the Marine and Coastal Access Act (2009) for proposals within the area of the English territorial sea capable of affecting, other than insignificantly, any of the protected features of a Marine Conservation Zone (MCZ) or any ecological or geomorphological process on which the conservation of any protected feature of an MCZ is (wholly or in part) dependent.

Natural England engagement in the six NSIP stages

Pre-application

Natural England is a prescribed consultee under the 2008 Act and secondary legislation made under it. Its input is vital in the pre-application process in cases where there are likely to be significant impacts on the natural environment and desirable where there are opportunities to deliver environmental net gain. Natural England encourages applicants to begin pre-application consultations at the earliest possible time and with as much detail as possible.

Early engagement with Natural England is very important because Natural England's advice (including on appropriate surveys and investigations) will enable applicants to give appropriate consideration to the environmental effects of an NSIP as the application is developing. The applicant has a duty to have regard to any consultation responses it receives[7] under sections 42, 47 and 48 of the 2008 Act. Matters concerning EIA, HRA, SSSIs and wildlife licensing should all be addressed during pre-application and are set out in more detail below.

Natural England encourages Applicants to take advantage of the **Discretionary Advice Service (DAS)** which is offered to provide non-statutory advice related to development proposals. We will not usually charge for s42 consultations and during Examination. Our terms and conditions can be downloaded from the above link.

To assist the applicant during the pre-application stage Natural England will continue to work closely with fellow Defra statutory environmental bodies (e.g. the Environment Agency or the MMO) when considering issues which cut across our various statutory responsibilities.

Acceptance

If matters relating to Natural England's involvement are not fully resolved in the pre-application stages Natural England will still engage with developers during the acceptance stage and advice is likely to be chargeable (see above re DAS advice).

Pre-examination

If an application is accepted for examination and there are outstanding issues of importance to the natural environment, Natural England is likely to notify the Examining authority that it wishes to be an interested party[8] and provide detailed relevant representations. We will then be able to engage with applicants for example on new technical information.

[6] Natural England's advice should be sought by developers prior to them carrying out works on or affecting a SSSI and in the case of owners and occupiers there is a requirement to notify and gain consent, prior to carrying out, or allowing to be carried out, works on or affecting a SSSI.

[7] Section 49 of the Planning Act 2008.

[8] Under section 89(2A)((b) of the Planning Act 2008

Examination

Where Natural England engages in the examination we aim to focus on providing written representations to update our earlier relevant representations. We will seek only to engage in relevant hearings in those complex cases where we have significant outstanding issues of high risk to the environment.

Recommendation and Decision

Where matters are not all resolved at Examination the sponsoring Government Department may come to us with queries through written consultation. During the decision making period any consultation with Natural England must be made public.

Post Decision

Natural England will engage in post consent activities such as where requirements are discharged or where we are consulted on variations, permits and licences. We will discuss cost recovery arrangements needed for this stage preferably before application.

Environmental Impact Assessment

Natural England has a statutory role as a consultation body under the Infrastructure Planning (Environmental Impact Assessment) Regulations 2017. Where an applicant has requested a scoping opinion from the Inspectorate in relation to a proposed EIA development, Natural England will be consulted by the Inspectorate in relation to the information they consider should be included in the environmental statement.[9]

Part 1 of Advice Note 11, and Inspectorate Advice Notes 3, 7 and 9, set out detailed advice on the EIA under the PA2008 regime and emphasise the importance of early consultation with Natural England.

Habitats Regulations Assessment

The Habitats Regulations place a responsibility on competent authorities (including the relevant Secretary of State where they are a 'decision maker') to consult the appropriate nature conservation bodies when carrying out an 'appropriate assessment' of the implications of an application for European sites[10]. For European sites in England, this means Natural England must be consulted and provide advice to the relevant competent authority if an NSIP is likely to have a significant effect on them. The Examining authority (ExA) will ask Natural England to provide its comments on the evidence that arises during the examination; the comments will be incorporated in or referred to in a Report on the Implications for European Sites (RIES) that will be submitted to the Secretary of State with the ExA's Report and Recommendation.

[9] Regulation 8 of the Infrastructure Planning (Environmental Impact Assessment) Regulations 2009 (as amended).

[10] The term 'European site' is formally defined in regulation 8 of the Habitats Regulations. It includes Special Protection Areas (SPA), which are classified under the Birds Directive, and Sites of Community Importance (SCIs) and Special Areas of Conservation (SAC), which are designated under the Habitats Directive. SACs and SPAs in marine or intertidal areas are European Marine Sites. Outside English territorial waters, SACs are designated as European Offshore Marine Sites (EOMS). Together they form the European Union-wide **Natura 2000** network. A full list of UK sites is available at http://jncc.defra.gov.uk/page-4 . SPA, SAC, EOMS, listed Ramsar sites, candidate SAC and proposed SPA are all referred to as 'European sites'. Note, however that a distinction is made between European Sites and European Offshore Marine Sites in the wording of the Habitats Regulations.

The Inspectorate strongly encourages the earliest possible liaison between applicants and Natural England, particularly where a European and/or Ramsar site or European Protected Species, or other protected species, may be affected by a proposed development. This will allow any relevant issues to be identified and if possible resolved at the pre-application stage. The Inspectorate has issued their own advice on Habitats Regulations Assessments here.

For projects which may affect a European Site, applicants for NSIPs located in England, or both England and Wales, are able to request the agreement of Evidence Plans with Natural England The aim is to set out the evidence needed in order to satisfy the legislative requirements and to agree how it will be collected and assessed. The aim of this approach is to ensure there is sufficient information available to enable the relevant Secretary of State to undertake a Habitats Regulations Assessment (including any appropriate assessment) it required. This approach (if followed) can reduce uncertainty at acceptance and save time in the examination. Natural England's experience is that this is a very valuable process, especially when started at an early stage in project design. It will engage with applicants in this way via the DAS. More information can be found here and applicants should approach Natural England if they are interested in agreeing an Evidence Plan.

Sites of Special Scientific Interest (SSSIs)

In relation to applications where there may be potential impacts on SSSIs both the Secretary of State and Natural England have duties under the WCA.

Under s.28(I), the Secretary of State or minister must notify Natural England before authorising the carrying out of operations likely to damage the special interest features of a SSSI. Under these circumstances 28 days must elapse before deciding whether to grant consent and the Secretary of State must take into account any advice received from Natural England, including advice on attaching conditions to the consent.

Natural England will give specific advice to applicants on the effects of the proposal on the special interest of any affected SSSIs when all the information is provided by applicants. It can also provide generic guidance and help on survey requirements. Applicants should seek to agree DCO requirements with Natural England, which may include measures for the protection of the SSSI, before the application is submitted to the Inspectorate. Natural England can provide advice to the decision-maker on any effects on the special interest of SSSIs.

Wildlife Licensing

Licensing of Protected Species

For all licensing matters, applicants should consult Natural England's published guidance for the relevant species and decide whether a mitigation licence is required. Applicants are able to use Natural England's charged Pre-submission Screening Service (PSS) for review of a draft wildlife licence application. This service can be used to receive early advice on all 3 licensing tests[11] (in relation to European protected species), before a Development Consent Order is granted. This service also extends to other protected species (such as badger, water vole), protected by domestic wildlife legislation.

[11] These tests are that there is "no satisfactory alternative" (Regulation 53 (9) (a) of the Habitats Regulations), that the activity authorised will not be "detrimental to the maintenance of the population of the species concerned at a favourable conservation status in their natural range" (Regulation 53 (9) (b) and that the licence is for a purpose specified in Regulation 53(1) which includes for "imperative reasons of overriding public interest including those of a social or economic nature and beneficial consequences of primary importance for the environment."

This early assessment is undertaken so that the decision-maker under the PA2008 can have confidence that Natural England, as the statutory licensing authority, has considered the appropriate issues relating to protected species. In order to do this, Natural England will conduct a review, based on a **full draft licence application, in advance of** the formal submission of the NSIP application to the Inspectorate. The steps to be followed when submitting a draft licence application to Natural England, or for early engagement on protected species licensing matters are set out in Appendix I of this note.

Following the review of the draft licence application, Natural England will either: provide a Letter of No Impediment (LONI), explaining that based on the information reviewed to date, that it sees no impediment to a licence being granted in the future should the DCO be issued; or if there are licensing issues to address, these will be set out in writing for the applicant to resolve. Only when all matters are resolved, following review of a subsequent draft licence application, can a LONI be issued. Any LONI will be sent to the applicant to provide within the application for examination. Natural England will copy any correspondence to the Inspectorate.

The ExA will wish to be in a position by the end of the examination to report to the Secretary of State on the likelihood of any necessary protected species licence being obtained, Natural England encourages all applicants to enter into early discussion in pre-application to ensure that where possible LONIs can be submitted at the application stage. Advice from Natural England under DAS, early on in the process, can help to scope out or refine protected species issues well before a draft wildlife licence application is prepared.

Strategic protected species licensing

Where strategic approaches such as district licensing for great crested newts are used a LONI will not be required. The developer will need to provide evidence to the ExA on how and where this approach has been used in relation to the proposal which may include a quotation from Natural England. A certificate will be issued by the habitat delivery body when compensation habitats are available.

Designated Landscapes

As the statutory landscape advisor and statutory consultee on EIA development, Natural England provides advice to applicants and the ExA on the landscape impacts of NSIP proposals on designated landscapes (National Parks, the Broads, and Areas of Outstanding Natural Beauty). Natural England is not able to comment on wider landscape and seascape issues and applicants should obtain advice from local planning authorities and other local bodies. Its advice will focus on the impacts of NSIP proposals on the statutory purposes of these areas to conserve and enhance natural beauty. This will include consideration of the landscape and visual impacts on the setting of these areas.

National Parks, the Broads and AONBs are designated because they represent our finest landscapes and as such are highly sensitive to the impacts of new development. The National Planning Policy Framework and National Policy Statements set out the policy position as it relates to development within a National Park or AONB. There is a presumption that development within these areas will be refused other than in exceptional circumstances and where it can be demonstrated to be in the public interest. Major development should be assessed against certain criteria including the need for the development within the designated area and the scope for moderating its effect on the environment. Natural England will provide advice to help the determining authority apply this ‘test’. Applicants should submit sufficient

information to inform this assessment.

Soils and best and most versatile agricultural land

Where significant development of agricultural land is demonstrated to be necessary, applicants should seek to use areas of poorer quality land in preference to that of higher quality, and protect soils during development. Natural England is the statutory consultee on certain development proposals affecting agricultural land which lead to (or are likely to lead to) the loss of 20 hectares or more of best and most versatile agricultural land (grades 1, 2 and 3a in the Agricultural Land Classification system) . In giving advice, Natural England will take into account the type of development, its likely long term effects and impact on soils. Under Schedule 5 of the Town and Country Planning Act 1990 (as amended) Natural England is also the statutory consultee for minerals and waste development where restoration to agriculture is proposed. Natural England advises on whether agriculture is an appropriate afteruse; whether soil management and restoration proposals meet the required standards and aftercare conditions are appropriate.

How to contact Natural England

All NSIP consultations for Natural England should be addressed to Natural England's casework hubs at consultations@naturalengland.org.uk. Applicants should email eps.mitigation@naturalengland.org.uk if they do not have a nominated case officer for protected species licensing issues. If for any reason it is not possible to consult electronically, consultations should be sent to the postal address below:

Natural England Consultation Service
Hornbeam House
Electra Way
Crewe Business Park
Crewe Cheshire CW1 6GJ

Appendix I: NSIPs and Protected European Species Licensing Issues

This advice has been prepared to help developers and developers' consultant ecologists understand the process for engaging with Natural England about Nationally Significant Infrastructure Projects (NSIPs) and issues relating to European Protected Species (EPS). Please note that this advice will be kept under review and may from time to time be amended.

Whilst this advice is primarily aimed at developments where EPS are affected (i.e. those species listed under the Conservation of Habitats and Species Regulations (2017), the procedure as set out below also applies to NSIPs involving protected species not covered by European legislation (e.g. *badgers, water voles, native white-clawed crayfish, Romans snails etc.).*
Please also note that any reference to the '3 licensing tests' below is not applicable to those species.

References to 'you' below should be taken to refer to 'developer and/or developer's consultant ecologist' as appropriate.

Introduction

For NSIPs which involve EPS, Natural England offers services, some of which are chargeable (see Annex A), to provide you with early advice and opinion on your protected species proposals in relation to all three licensing tests[12] without a planning consent needing to be in place. This is undertaken so that the examining authority for NSIP applications under the PA2008, can have confidence that Natural England, as the relevant licensing authority, has considered the impacts relating to protected species in order to make a recommendation to the relevant Secretary of State.

In order to do this, Natural England needs to conduct an assessment, based on a **full draft mitigation licence application,** in advance of the formal submission to the Inspectorate. The steps to be followed when submitting the appropriate information, in respect of an NSIP project which has the potential to affect EPS, are set out below and in the Flow Chart on page 7.

Key message for NSIP developers and their ecological consultants:

[12] The three tests:

- the activity must be for a certain purpose (for example, for scientific research or in the public interest)
- there must be no satisfactory alternative that will cause less harm to the species
- the activity must not harm the long-term conservation status of the species (you may need to create new habitats to offset any damage)

Requirements for a European Protected Species Licence

Natural England strongly advises that developers engage at the earliest possible opportunity with Natural England should protected species licences be required.

We recommend that a draft licence application is generally* submitted at the pre- application stage to assist the examination process. You should be aware that if you choose to submit your DCO application to the Inspectorate without having resolved matters relating to licensing first then there is a significant risk that these issues may prevent your application proceeding past the application or examination stage. Please note that, in cases where a licence is required and the licensing team has not been appropriately consulted, in accordance with the process outlined below, Natural England cannot be held responsible for any delays experienced with regards to the progress of your DCO application or if the application is unsuccessful as a result of outstanding licensing issues.

**Note for large linear schemes (pipelines, cables, etc) from our experience it is preferable for the route to have been agreed before submitting a draft great crested newt licence application so the draft information, particularly in relation to survey and impacts, are clear and understood. Submission should still be at the earliest opportunity to ensure that any issues which require resolution are dealt with in a timely fashion well in advance of the examination stage.*

Step 1 Is a licence required? Informal engagement with Natural England

You should consult Natural England's published guidance for the relevant species and decide whether a mitigation licence is required. If you decide that a licence is required you should start to develop the mitigation scheme which should follow Natural England's published guidance for the relevant species. Although optional, to avoid any unforeseen problems arising further along in the process, you are encouraged to engage as early as possible with Natural England. If you do not have a nominated case officer at Natural England for your project then for licensing purposes, please use the following email contact:

- eps.mitigation@naturalengland.org.uk: mark your email 'NSIP – informal engagement' -*with the name of the NSIP project, and the species concerned, in the email subject header*. This will enable Natural England to coordinate, advise and oversee NSIP licensing issues through the correct channels. Once an application is accepted it will be overseen by
- the relevant Natural England Area Team.

Natural England can provide informal advice about licensing issues at any time or, through free general advice at the initial stage can outline what information is needed to facilitate this informal pre-application discussion further.

Natural England aims to provide this general or initial advice on licensing requirements over the telephone within five working days. If, however detailed pre-licence application advice is requested (e.g. for a teleconference or meeting), this will be provided through DAS in accordance with published timescales

For particularly complex cases, when the Natural England team is experiencing high workloads, or where a site visit is considered necessary for advice to be given, Natural England will contact you to discuss when it will be possible to provide a view on the case.

At this stage in the process, no full assessment of the Method Statement[13] will take place. Depending on the level of risk or opportunities presented through the mitigation, a teleconference or face to face meeting may be appropriate to discuss matters in detail.

Step 2 – Draft licence application submitted to Natural England

For Natural England's officer to provide pre-submission screening advice on the acceptability of the proposed licence application and mitigation, applicants should**,** as soon as the proposals are sufficiently advanced and that the mitigation proposals take account of the final design,:

- Prepare a full draft licence application including :
    - An application form,
    - Method Statement and maps/figures together with a proposed Work Schedule (which should be Specific, Measurable, Achievable, Realistic and Time-limited ("SMART"), and
    - Reasoned Statement[14].
- Email it to eps.mitigation@naturalengland.org.uk, marked 'NSIP – Draft licence application' with the NSIP name and species concerned included in the email header.

When preparing the application form documents for submission, the guidance on naming files and using folder structures as set out in the 'Key message' in section 16 of the 'How to get a licence' document should be followed.

When submitting draft application documents by paper or electronically, the guidance within the application form and the 'How to get a licence' document (section 16) should be followed (eg documents over 5MB in size should be submitted to Natural England on CD rather than by email). A link to Natural England's file compression guidance is provided below. Applicants are advised to reduce the size of their application pack when submitting by email as far as possible.

Please note that, in terms of the three licensing tests set out under Part 5 of the Conservation of Habitats and Species Regulations (2010) as amended ("Habitats Regulations"), it will not be possible for Natural England to consider that these tests have been fully met, in respect of any NSIP, until the Secretary of State has granted the DCO. However, a full assessment of your draft application, including the Reasoned Statement and supporting evidence, will be undertaken in advance of the DCO being granted, in accordance with the terms of Natural England's pre-submission screening

[13] A Method Statement shows what impact an activity will have on protected species and whether their favourable conservation status will be maintained.

[14] This document is used by our EPS Advisers to assess whether the Purpose and No Satisfactory Alternative tests have been met. It requires you to set out your views and provide evidence to demonstrate that the proposed activity meets one of the prescribed purposes. In addition, evidence is required to demonstrate that there is no satisfactory alternative to undertaking the activity, with lesser impacts on the species, as proposed in the licence application.

advice, so as to determine whether the appropriate level of detail has otherwise been provided (please see Flow Chart at the end of this document).

Please ensure that the necessary documentary evidence, which supports the statements made within the Reasoned Statement in respect of the Purpose and No Satisfactory Alternative Tests, has been included. For projects of this scale, even though the required consents will not yet have been obtained, Natural England still expects there to be a sufficient amount of supporting evidence available (e.g. reports, studies etc.) which demonstrate the need for the development and other alternatives which have been considered and subsequently discounted as being less satisfactory. Please note that we will be unable to issue the 'letter of no impediment' until the appropriate level of information has been provided in respect of all three tests.

Step 3 – '**Letter of no impediment' or 'further information request' issued from Natural England**
Within 30 working days, Natural England will either issue:

- 'a letter of no impediment' stating that it is satisfied in principle, in so far as it can make a judgement on the information reviewed, that the proposals presented comply with the 3 licensing tests, or
- a letter outlining why we believe the proposals currently do not meet licensing requirements and what further information is required. If further information is required, this is likely to result in the need for further advisory services under the pre-submission screening agreement of the revised draft licence application. It should be noted that time taken by you to provide any amended/enhanced/new information does not count towards the 30 working day customer standard target.

Correspondence on the draft applications and advice given will be copied to the Inspectorate.

The **'letter of no impediment'** will set out Natural England's advice against the three statutory licensing tests under Part 5 of the Habitats Regulations. You can use this letter to support your application to the Inspectorate; it will also be sent by Natural England to the Inspectorate. The letter will make clear that, on the basis of the species information and proposals presented to date, Natural England is satisfied in principle that the licensing tests are likely to be met when a formal application is submitted (on the basis that the information/evidence provided within the application remains the same), subject to the DCO being granted by the Secretary of State and subject to the caveats listed under clause 4 of the PSS terms and conditions (see link below). The letter will also draw attention to the fact that ecological conditions on the site may change over time. It is your (the developer's) responsibility to maintain sufficiently up to date survey information which is then made available to Natural England (along with any resulting amendments to the draft licence application) and the Inspectorate so that there is no delay in issuing the licence once the Secretary of State has granted the DCO.

Step 4 – Submission of the NSIP application to PINs for a Development Consent Order

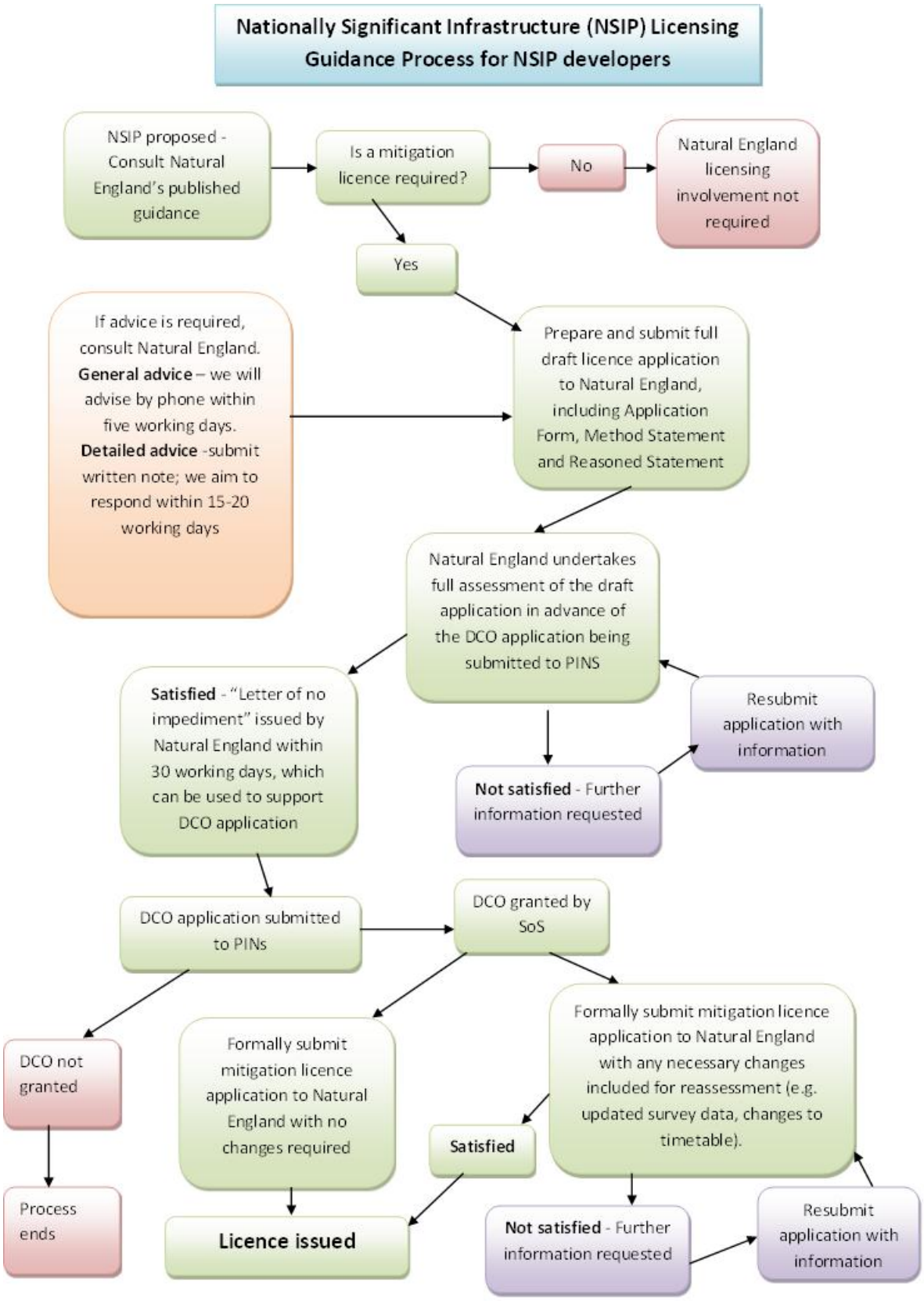
After you have submitted the DCO application to the Inspectorate, along with the 'letter of no impediment' and associated mitigation proposals you should keep Natural England informed of progress to ensure that we remain aware of the likely timeframes so that we know when to expect the official application and can undertake a timely final mitigation licence decision. It is possible that the timetable of activities (which forms a legally enforceable part of the European Protected Species licence – see regulation 53(8)(c)(ii) of the Habitats Regulations) will require a final update if there has been any slippage in the agreed timings (see Step 5).

Step 5 - Natural England grant of a mitigation licence following the Development Consent Order being issued (note this step is not chargeable).

Once the DCO has been granted, you should formally submit the mitigation licence application to Natural England (following the submission process outlined in Step 2)), Natural England will grant a licence, provided the proposals and the situation on site either:

- Remain the same and the work schedule is still SMART (Specific, Measurable, Achievable, Realistic and Time-limited), or
- Have been suitably adjusted to enable Natural England to confirm that the mitigation proposals remain adequate (eg timings in the work schedule may change), or
- Take account of any further survey requirements, and reassessment of impacts etc, resulting from a significant delay between the issue of the 'letter of no impediment' and the DCO decision.

Please be aware that if changes are made to proposals or timings which do not enable us to meet the three tests we will issue a letter outlining why the proposals are not acceptable and what further information is required. These issues would need addressed before a licence can be granted. This will also be sent to the Inspectorate to keep them informed of any advice given by Licensing to you (the developer). Natural England does not expect any significant changes to be made to proposals agreed in principle at this stage.



Useful links for preparing a draft application:

Please note that our advice is regularly updated. It is therefore advisable to use the latest versions available on Wildlife Management and Licensing .GOV.UK web-pages. Core guidance is listed below:

- 'How to Get a Licence'. This gives an overview of the licensing process.
- https://www.gov.uk/environmental-management/wildlife-habitat-conservation gives useful links to species licence information
- Application forms web link: https://www.gov.uk/wildlife-licences
- Pre-submission screening service



Kenneth Pye Associates Ltd.

Scientific Research, Consultancy and Investigations

Coastal Processes and Morphological Evolution of the Minsmere Reserve and Surrounding Area, Suffolk

Report prepared for the RSPB (East Anglia Office)

by

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and

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February 2005

EXTERNAL RESEARCH REPORT ER511

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1. Introduction

This coastal processes report has been produced at the request of the RSPB (East Anglia Office), to provide a better understanding of the historical evolution of the Minsmere site, which will aid assessment of different options which are being considered for the future management of this section of the Suffolk coast.

Minsmere is located approximately 30 km south of Lowestoft, in an area which is of both national and international environmental importance (Figures 1 and 2). Most of the site is designated as a Site of Special Scientific Interest (SSSI), and the area to the north of the New Cut drain forms part of the Minsmere to Walberswick Heaths and Marshes Special Area of Conservation (SAC), the Minsmere to Walberswick Special Protection Area (SPA), and the Minsmere to Walberswick Ramsar Site. The RSPB has leased land in the area since 1947 and purchased the reserve in 1977. Today Minsmere is one of the RSPB's 'flagship' reserves with more than 80,000 visitors each year. It contains a diverse assemblage of wetland and other habitats, including extensive freshwater reed beds, and is home to up to 90 species of breeding birds including bitterns, avocets, marsh harrier and marsh tit (Welch, 2003). The reed beds cover 179 ha and form the third largest block of this habitat type in England.

Although reclamations have occurred on the site since the twelfth century, Minsmere was an estuary until the eighteenth century. The development of a shingle beach and sand dune barrier blocked the seaward end of the estuary, and an extensive drainage system with a sea sluice gate were installed in the early part of the nineteenth century. As part of these works, an artificial clay bank was also built behind the beach and dune ridge, which acted as a second line of the coastal defences for the reclaimed marsh behind. Adjoining the Minsmere Reserve to the south are the Sizewell A and B nuclear power stations whose defences consist virtually entirely of the beach and dunes to seaward, although the power stations were constructed on slightly higher ground, and a large artificial bank and been built behind the beach to the north of the site.

The Minsmere frontage and adjoining sectors of coast are experiencing long-term net erosion which historically has occurred at an average rate of *c.* 0.5 m yr^{-1}, locally exceeding

1.0 m yr^{-1}. Based on future climate change predictions, sea level is expected to rise on the Suffolk coast and rates of coastal erosion may also be expected to increase, creating pressure on the flood defences. While recognizing that it may not be possible to maintain the Minsmere Reserve in its present form indefinitely, the RSPB would wish to maintain its essential character, notably the freshwater habitats, for as long as is practically possible, and wish to ensure that all available options are given thorough consideration.

Sources of information

In preparing this report the following sources of information have been used:

(i) historical maps of the coastline (1575 to present)
(ii) historical air photographs (1940 to 2003)
(iii) historical Admiralty Charts (1968 to 2004)
(iv) beach and bathymetric profile data (1991 to 2003)
(v) lidar topographic data (1999)
(vi) historical records of tide levels (1972 to present)
(vii) historical records of mean sea level (1955 to 2003)
(viii) published and unpublished scientific literature
(ix) reconnaissance site visits in June and December 2004
(x) field data collected during earlier studies of the Suffolk coast since 1990

The unpublished literature considered includes preliminary documents prepared by, and on behalf of, the Environment Agency in relation to the proposed Minsmere coastal management strategy, and various reports prepared by Professor John Pethick relating to coastal processes and monitoring of shoreline changes around the Sizewell nuclear power stations.

2. Overview of coastal processes on the Minsmere frontage

2.1 Tides

Tides on the Suffolk coast are semi-diurnal, with an average progression of approximately 30 minutes between successive tides. Minsmere is approximately midway between the Standard Ports at Lowestoft and Felixstowe. Predicted tidal data for these and secondary ports along this frontage are given in Table 1. The tidal range decreases considerably to the north of Felixstowe, reaching a minimum at Lowestoft with a mean spring tidal range of just 1.9 m. At Minsmere, the mean spring tidal range is 2.1 m. The level of high tide relative to OD reaches a minimum in the vicinity of Minsmere, where the level of predicted mean high water springs is *c.* 0.8 m OD and that of mean high water neaps *c.* 0.4 m OD (Table 1).

Figure 3 shows the annual average high and low tide levels recorded at Lowestoft over the period 1972 to 2003. Both show a general rising trend over this period. The average level of high tide rose by *c.* 13 cm (an average rate of 4.2 mm yr^{-1}), while the average level of low tides rose by *c.* 17 cm (an average rate of 5.5 mm yr^{-1}). Superimposed on this trend is an oscillation in the difference between mean high and low tide levels (i.e. tidal amplitude) caused by the lunar nodal tidal cycle, whereby the plane of the moon's orbit around the earth rotates over a period of 18.61 years. This causes the amplitude of semidiurnal tides to vary over the same timescale, with maxima in 1978, 1997 and 2015, and minima in 1969, 1987 and 2006. Annual average tidal amplitudes can be calculated by subtracted the two curves in Figure 3. The resulting curve (Figure 4) clearly shows the lunar nodal oscillation in annual tidal amplitude. The curves are not smooth due to periods when the recording gauge was not working, and random weather events which affect the timing and height of low and high water.

The flood tidal stream offshore from Minsmere runs almost directly southwards (*c.* 185°) down the coastline (Admiralty, 2004). Average flows reach a maximum 5 hours before high water, at 0.67 m s^{-1} on spring tides and 0.31 m s^{-1} on neap tides. The ebb tidal stream runs almost exactly northwards (*c.* 007°). Average flows reach a maximum 1 hour after high water, 0.72 m s^{-1} on spring tides and 0.36 m s^{-1} on neap tides. Modelled current residuals for the sea bed (Halcrow, 2000d) are very small, although directed southwards. The maximum residual

flow reaches 0.05 m s^{-1} over Dunwich Bank, and near the shore along the Dunwich Cliffs. Elsewhere residual flows are less than 0.05 m s^{-1}.

2.2 Impact of storm surges

The Suffolk coast is frequently affected by surges which change the height and duration of predicted tidal levels; both positive and negative surges can occur. Positive surges (higher than expected tides) are most frequently associated with slow moving depressions in the North Sea which draw down a strong northerly or north-easterly airflow. This has the effect of 'piling' water against the coast. Low pressure also results in a rise in the mean surface water level. Since the astronomical tidal range is so small along this part of the coastline, surges can have a large impact in the tidal level at the coast.

Large surges combined with high tides occurred in 1817, 1883, 1897, 1912, 1928, 1938, 1949, 1953, 1976 and 1978 and caused regional damage to flood defences, agricultural land and property (Corkan, 1950; Jensen, 1953; Steers, 1953; Grove, 1953; Robinson, 1953; Rossiter, 1954; Pollard, 1978; Summers, 1978; Steers *et al.* 1979). There are also numerous historical references to major storm surge floods in earlier periods (Lamb, 1991), although their precise effects on the coast are poorly documented. The best documented surges are those of 1978 and 1953. The storm surge of 11 January 1978 increased the height of predicted high tide at Southwold by about 1.0 m, resulting in an actual high tide level of 2.00 m OD (Table 1) (Steers et al., 1979). Another a surge on 3 January 1976 produced even higher levels, the tide at Southwold reaching 2.50 m OD. However, the surge of 31 January 1953 produced the largest surge recorded on this coastline, and resulted in the highest high tide levels (3.50 m OD at Southwold and 3.78 m OD at Aldeburgh). Although no records exist, the surge of 1938 was possibly as severe as that in 1953. Historical aerial photographs (see later in this report) show a large washover fan at Minsmere which was caused by a breach in the defences during this storm (Steers, 1951).

Since 1978 damage from storm surges has not occurred on a major scale, partly due to meteorological factors and partly due to improvements in sea defences. Figures 5 and 6 show the monthly maximum storm surges, and the monthly extreme tidal heights recorded at

Lowestoft and Felixstowe between 1990 and 2004 (data are from the National Tidal and Sea Level Facility). These show that surges in excess of 1 m occur relatively frequently along the coast. The most recent extreme tide occurred on the night of 14 February 2005, when the mean level reached 2.0 m OD at Lowestoft. This tide overtopped the dune ridge at two locations on the northern Minsmere frontage over a length of a few metres. The dune ridge was also overtopped at this location on 14 to 15 December 2003, when the mean level reached 2.2 m OD at Lowestoft. The previous occasion when a tide exceeded this height was on 29 October 1996, when the tide reached 2.3 m OD at Lowestoft. There were 5 previous occasions over the period when the tide exceeded the level in December 2003, and the tide exceeded a height of 2.0 m OD on a total of 15 occasions. The maximum tide levels during this period of record occurred on 21 February 1993, when the tide reached 2.6 m OD at Lowestoft. This storm surge caused damage and some overtopping of defences at many locations along the east coast of England.

2.3 Winds

The nearest station for which long-term annual average wind data are available is at Hemsby, near Great Yarmouth (Figure 7). Records from 1981 to 2000 show that the prevailing winds blow from the southwest and the calculated resultant is directed towards the northeast. While winds from all directions are able to influence aeolian sand transport along the Minsmere shoreline, transport of sand from the beach can only occur when winds blow from the northeasterly or southeasterly quadrants.

2.4 Waves

No long-term measured inshore wave records are available for this coastline, although wave recorders have been deployed at several locations for short periods at varying times: the Institute of Oceanographic Sciences installed wave recorders along the coast and on the offshore banks between 1975 and 1979; the Environment Agency recorded waves at the entrance to the Blyth Estuary in 1995-1996; and CEFAS installed a gauge offshore from Sizewell in 2003. The Meteorological Office UK Waters Wave Model produces hindcast

wave data for the period since 1986, based on the recorded distributions of surface winds. Calculations for a point 48 km east of Dunwich, based on this hindcast data for the period 1986-1999, are given in Table 2, and an offshore wave rose is shown in Figure 8. The data show that there is a clear bimodality, most waves approaching from two opposing directions between north and northeast (345° to 045°) and between south and southwest (165° to 225°).

Wave energy can be classified as moderate. 37.7% of all waves are less than 1 m high, and 76% of all waves are less than 2 m high. The highest waves approach from the north to northeast, which is the direction with the longest fetch. There is therefore a potential wave-generated current residual in a southward direction, although the rate is small at less than 500 kN s^{-1} (Figure 9) (Anglian Water, 1988a). Halcrow (2000d) concluded that, even during storm conditions, wave induced currents are so small that their role in promoting sediment transport may be neglected.

Predicted extreme wave heights are displayed in Table 3. The highest waves are predicted to approach the coast from the north and northeast, with 1 in 100 year heights predicted to be 7.3 to 7.8 m. From other directions the 1 in 100 year heights range from 5.4 to 6.7 m. The highest waves predicted to occur during any one year are 5.0 m, from a northerly direction, and no more than 4.3 m from any other direction.

Local inshore wave heights, period and approach angle are strongly controlled by the morphology of the coastline, and by the offshore bathymetry. Since this part of the Suffolk coast is one of the most easterly in the UK, there is little protection from adjoining headlands, although the Southwold to Lowestoft coastline provides limited protection to the north.

Previous studies (e.g. Robinson, 1980; Carr, 1981) have emphasized the importance of Dunwich and Sizewell Banks for reducing the wave energy which reaches the coast. Indeed, Robinson concluded that a reduction in cliff erosion rates observed in the last fifty years can be attributed to the growth of Sizewell and Dunwich Banks which have reduced wave energy at the shoreline. He also showed that the northward extension of the bank altered the angle of wave approach, and pattern of wave focussing, at the coast. Historical tracer experiments (Lees, 1981, 1983) have also shown that sediment is moved landwards by wave action, resulting in a landward shift in the position of the banks, during stormy conditions. This

migration will absorb some wave energy, with a subsequent reduction in the energy which reaches the coastline behind the banks. As the banks move closer to the shore, there is less opportunity for refracted waves passing over the bank crests to regenerate before they hit the shoreline.

The effect of the banks on wave activity was investigated by the Institute of Oceanographic Sciences (Deacon Laboratory, Surrey) between 1975 to 1979, when three wave recorders were installed close to the shore at Southwold, Dunwich and Aldeburgh (Fortnum and Hardcastle, 1979). No significant reduction in wave energy could be detected over this period at Dunwich, despite the location behind the offshore banks. However, later studies using wave rider buoys placed on either side of Sizewell Bank between 1978 and 1979 showed that large waves (> 2.2 m in height) broke on the offshore banks during certain conditions of wave height and direction, thereby providing some protection to the coastline (Carr, 1981).

Computer modelling by Halcrow (2000d) suggested that the influence of the banks is not very important in terms of the energy which reaches the shoreline during typical or moderate wave conditions. The computer model was initially run for a spring tide with waves approaching shore normal (105°) and with the most recent available bathymetry. The model was then re-run removing Dunwich and Sizewell Banks entirely from the bathymetry. Results showed that the banks locally reduce wave heights over the banks by up to 0.5 m, although the effect of the banks on wave heights at the shoreline was predicted to be negligible. However, the banks were shown to focus wave energy at the shoreline, causing diffraction of wave energy and a slight change in wave direction at the shoreline.

The Halcrow modelling considered only one set of wave conditions with a fixed angle of approach. It remains to be demonstrated that such models are capable of accurately representing the complex three-dimensional and time-dependent interaction between the offshore banks and processes at the shoreline, including cliff erosion. While the banks may have little influence on waves at the shoreline during typical weather conditions, they are likely to be far more important in sheltering the coast during storms, or when waves approach from a particular direction. The northward growth of Dunwich Bank over the last 50 to 60 years (see later sections) is highly likely to have had a significant effect in extending a degree of storm wave protection northwards from Minsmere towards Dunwich.

2.5 Geology

The Suffolk coastline lies on the western margin of the North Sea Basin. During the late Pliocene and early Quaternary, three sequences of clastic strata were deposited in the area: the wholly marine Coralline Crag, of Pliocene Age, and the younger Red Crag and Norwich Crag (Hamblin *et al.*, 1997). Several separate units have been identified within the Norwich Crag, including the Chillesford Sand, Easton Bavents & Chillesford Clays, and the Westleton Beds. These units are largely composed of marine sands but include some gravels and clays. The Westleton Beds are particularly coarse, their previous name being the 'Westleton Sands and Shingle', or 'Pebbly Sands' (Prestwich, 1890). Variations in grain size throughout the Norwich Crag formation have been related to changes in relative sea levels during the Pleistocene (e.g., Hey, 1967; Sinclair, 1999).

Above the Norwich Crag lies boulder clay associated with the Lowestoft Till, and then deposits of sand and gravel derived during the Holocene. Much of the nearshore zone is composed of Holocene sands and gravels, while Dunwich and Sizewell Banks further offshore contain predominantly sand (Figure 10). Mud predominates offshore south of Thorpeness, while elsewhere the sea bed is formed by the Norwich Crag.

2.6 Sediment sources and transport patterns

Minsmere lies within Coastal Processes Sub-Cell 3c, which extends from Lowestoft to Harwich (Halcrow, 1998). The general net sediment transport trend in this sub-cell is southwards (Motyka and Beven, 1987). There is general agreement that alongshore sediment input from the north is very small at the present day, as is onshore movement of sediment from deeper water. The most significant input of sediment to the nearshore zone within the Sub-Cell is from the erosion of soft coastal cliffs, notably at Pakefield, Benacre, Covehithe, Easton Wood, Easton Bavents, Dunwich and Minsmere Cliffs. The Dunwich and Minsmere cliffs are mainly composed of Norwich Crag group sediments, which have a high sand content (on average 90%; British Geological Survey, 1996), although gravel constitutes up to 60% in some areas, and some glauconitic sand layers contain up to 15% mud. Further south, the

Sizewell Cliffs are composed of Chillesford Sand, which contains very little gravel or mud (Figure 10).

Shingle is most abundant nearest to the source of supply, i.e. the eroding cliffs (Carr, 1981). Sand is preferentially transported southwards along the coast, relative to gravel. The nesses at Kessingland, Southwold and Thorpeness act as important local net sediment sinks, especially for coarser sediment, with the sub-cell. There has been some argument in the literature as to whether these nesses also act as sinks for sediment transported from offshore (Robinson, 1980), or are locations where sediment is transported to the offshore zone (McCave, 1978; Carr, 1981). At Thorpeness, the consensus of opinion is that while shingle tends to be retained on the beach, there is a net transfer of sand seaward towards the nearshore and offshore banks. Lees (1983) suggested that a tidal residual eddy may exist at the promontory of Thorpeness, which has remained essentially fixed in position by a relatively resistant outcrop of Coralline Crag. Sediment is then transported offshore to Sizewell Bank, and then northward to Dunwich Bank. Lees' studies showed that this northward transport is via the suspended sediment mode, and may even be responsible for the formation of Sizewell and Dunwich Banks. Recent computer modelling (Halcrow, 2000c) has also demonstrated the potential for sediment to move offshore from Thorpeness into the banks during storms, although sediment transport is greatly reduced under average wave conditions.

Several studies have attempted to quantify a sediment budget for the coastline between Lowestoft and Thorpeness (e.g., Vincent, 1979; Carr, 1981; Clayton *et al.*, 1983; Halcrow, 2000c). Estimates by Vincent (1979) and Clayton *et al.* (1983) used historical records of cliff recession rates for the previous 100 years, and measurements of beach volume change from the 1960s and 1970s. Halcrow used estimated cliff recession rates from short-term beach profile data and computer modelling to estimate longshore sediment transport rates. Since these studies assessed change on very different timescales, the estimates of the sediment budget vary considerably. Estimates of the input of sediment from the Minsmere and Dunwich cliffs vary from 40,000 m^3 yr^{-1} (Vincent, 1979) to 12,000 m^3 yr^{-1} (Halcrow, 2000c). Vincent (1979) estimated the southward net littoral drift of sediment at Dunwich to be 65,000 m^3 yr^{-1}, while Halcrow (2000c) calculated a maximum figure of 12,000 m^3 yr^{-1} for a shorter, more recent period.

Carr (1981) assessed changes in sediment volume over the much longer timescale to 1867 to 1965, based largely on map evidence. Carr's calculations can be summarised as follows:

			Change (m^3)
Losses from cliffs:	Easton Bavents		4,366,300
	Dunwich-Minsmere		6,310,900
	Thorpe Ness		235,700
Losses from nearshore:			7,750,000
		Total losses:	18,662,900
Gain in volume of Sizewell-Dunwich Banks:		Total gains:	21,921,900
Total volume of sediment lost from the 'shoreline' system:			3,259,000

Carr's calculations show that the erosion of material from the coastal cliffs is almost balanced by accretion on the Sizewell and Dunwich Banks. This suggests that the coastline between Southwold and Thorpeness behaves almost as a closed system, sediment derived from cliff erosion firstly being transported south to Thorpeness, and then offshore to Sizewell and Dunwich Banks. There appears to be a relatively small loss of sediment out of this section of coastline to the south of Thorpeness. Halcrow (2001a) estimated an annual drift to the south of Thorpeness of 10,000 m^3 yr^{-1}, which equates to roughly the same order of magnitude as the loss calculated by Carr.

2.7 Past sea level changes

Based on radiocarbon and sea level index point data compiled for the area around Great Yarmouth, relative sea level in East Anglia was *c.* 30 m lower than present at 9000 years BP (Figure 11) (Coles & Funnell, 1981; Shennan & Horton, 2002). Analysis of successive units of peats and estuarine clays in the Blyth Estuary have shown that there have been several changes between transgressive and regressive periods in the last 9000 years (Brew *et al.*, 1992). Regressive periods associated with peat formation occurred between 6750 and 6500 years BP, and between 4500 and 4300 years BP. Sandwiched between these periods were transgressive phases when estuarine clays were deposited.

Data for the last 4000 years are sparse, especially for Suffolk, but in northern East Anglia Shennan & Horton (2002) estimate that the average rate of relative sea level rise since 4000 years BP has been *c.* 0.61 mm yr^{-1}. There was probably a slight fall in relative sea level between *c.* 3000 and 2500 years ago, prior to the Romano-British transgression 2500 to 2000 years ago (Brew *et al.*, 2000). This was followed by a slight fall and then further transgression culminating around 1700 to 1600 BP. Sea level may have fallen again between 1400 and 1300 BP, but then been relatively high during the Medieval Warm Epoch between 900 and 600 BP. Slightly lower sea level is then suggested during the Little Ice Age, which reached maximum intensity between 1650 and 1750 AD (Hulme, 1994). Lamb (1995) suggested that the phases of higher relative sea levels (such as 1700 to 1600 BP and 800 to 700 BP) may have been associated with a relative warming of the atmosphere and a northwards defection of westerly storm tracks.

Historical tide gauge records at Lowestoft have indicated that there has been a general trend of rising sea level over the period 1972 to present. The average level of high tide rose by *c.* 13 cm (an average of 4.2 mm yr^{-1}), while the level of low tides rose by *c.* 17 cm (an average of 5.5 mm yr^{-1}). However, such records must be treated with some caution, as gauges are affected by variations in storminess, and by changing tidal prisms due to dredging operations near to ports. Shennan & Horton (2002) quote a figure for relative sea level rise at Lowestoft during the twentieth century of 1.81 ± 0.48 mm yr^{-1}, and as yet there is no clear evidence of significant geomorphological effects due to any recent acceleration in the past 40 to 50 years.

2.8 Past changes in wind-wave climate and storminess

Significant variations in weather and climate have taken place over the past 2000 years and, indeed, have been significant on a decadal time scale. These have affected sediment transport patterns, local sediment budgets and morphological responses. The most marked fluctuations were the Medieval Warm Period (1000 to 1200 AD) and the Little Ice Age (*c.* 1500 to 1850 AD). According to the model proposed by Lamb (1995) warming phases are associated with an increase in winds from the southwest, west and northwest, and cooling phases with a greater relative importance of winds from the north and northeast. If this model is correct, cooler periods such as the Little Ice Age should be associated with greater wave energy,

larger and more frequent storm surges, and greater coastal erosion and flooding. Under such conditions, north to south sediment drift along the Suffolk coast should be enhanced. Conversely, during periods of westerly dominance, there should be less general erosion and storm surge damage, and the coastline should experience relative stability. During periods of frequent storm erosion, large amounts of sediment re-enter the nearshore zone and are available for redistribution

As noted earlier in this report, long-term wind, wave and sediment transport data are lacking for this coast, as for most other parts of the United Kingdom. The causes of observed coastal changes therefore have to be inferred from imperfect proxy data, frequently relating to recording sites some distance away. Detailed coastal monitoring is a recent phenomenon, and even in East Anglia, which is fortunate in having one of the most comprehensive and enlightening coastal data capture systems in the country, the availability of systematic annual or seasonal data is restricted to the period since 1991.

Temporal changes in wind (and therefore wave) regime can to some extent be inferred from records of temperature and precipitation. A composite record of temperature variation in Central England extending back to 1659 is available (Manley, 1974, Hulme, 1994). This shows that mean annual temperature was higher than the long-term average (1659-1992) for short periods in the 18th century (1720-1740) and 19th century (1820-1830 and 1865-1875) and has consistently been above the long-term average between 1890 and the present. The coldest period was between 1640 and 1700. A similar long-term composite precipitation record for England and Wales exists for the period 1766 to 1991. This indicates peaks of wetter periods around 1766, 1790, 1825, 1875, 1925 and 1955, with drier periods centred around 1775, 1810, 1850, 1890, 1940, 1965 and 1985 (Wigley *et al.* 1984; Hulme, 1994). Although the relationships between temperature, precipitation and wind systems are not simple, there is a broad association between frequency / duration of westerly weather types (with dominance of SW, W and NW winds) and higher temperatures. Therefore, using this association, the late seventeenth, late eighteenth and early nineteenth centuries were periods when relatively strong north-south coastal sediment drift along the Suffolk coast could be expected to have prevailed. Conversely, the last 110 years should have been a period of weaker north-south sediment drift, especially after about 1925.

These broad trends might explain, at least partly, the rapid erosion of the Minsmere and Dunwich Cliffs observed from historical maps in the early nineteenth century, while cliff recession rates have reduced in recent decades. These trends are discussed further in the following section.

3. Overview of past geomorphological changes on the Minsmere frontage

3.1 Evidence from historical maps (1575 to 1976)

The earliest documented evidence of the geomorphology of the Suffolk coast dates back to the twelfth century (Steers, 1927). At this time, a number of small streams entered the sea near Kessingland (TM 538 845), Benacre (TM 533 829), Easton (TM 520 795), Dunwich (TM 480 707), Minsmere (TM 478 661), and Thorpeness (TM 472 590). Each was open to the sea and many served as havens (Suckling, 1846). On the Minsmere Estuary, Leiston Abbey was founded in 1182, and the monks began building clay embankments to reclaim areas of marshland.

In the thirteenth and fourteenth centuries, a series of major storms caused severe flooding along the whole east coast of England. These storms moved sand and gravel from offshore banks onto the coastline, and periodically blocked the entrances to the havens. Dunwich at this time was a thriving port, with a population of thousands, and 18 churches, chapels and monasteries. During a particularly bad storm in 1286, much of the town was destroyed (Bacon & Bacon, 1984). The town was an important port situated on the mouth of an estuary which served as the entrance to both the Dunwich River and the River Blyth. In the following decades there were severe problems with mud and sand bars developing in the estuary mouth. During a storm in 1328, the Dunwich haven became blocked completely by a shingle bar (Steers, 1927). Repeated attempts to form a cut through the shingle failed, and a new cut was ultimately made to the south of Southwold.

Severe flooding also occurred at Minsmere in 1347 and 1363. This prompted Robert de Ufford to relocate Leiston Abbey further inland, nearer to Leiston, leaving only the small chapel on an area of higher ground near the estuary mouth. The small village of Minsmere, which consisted of about 10 houses and a small church, was also flooded, being lost to the sea by the sixteenth century (Bacon & Bacon, 1984).

More recent changes along the coast can be dated by examination of historical maps (see summary in Table 4). Saxton's map of 1575 (Figure 12a) shows that, despite periodic blocking by shingle, the havens at Minsmere, Easton and Kessingland were open to the sea at

this time. Kirby's map of 1737 (Figure 12b) shows that the Minsmere River at this time followed a very similar course to the present New Cut, flowing slightly south of east from Dam Bridge, and entering the sea just to the north of the old chapel (at approximately the same position as the present sluice). Around this time, partially aided by a decline in the rate of relative sea level rise and/or storminess, shingle barriers began to form along the coast. To the north, the haven at Benacre was closed to the sea by 1737, and a large broad, Benacre Broad, was created behind the new beach barrier. The estuaries at Easton, Minsmere and Thorpeness also became blocked in the following decades. At Minsmere, a shingle barrier had formed by 1752, and had completely dammed the estuary by 1780 (Axell and Hosking, 1977). Hodskinson's map of 1783 (Figure 12c) shows that broads (or lagoons) were created in many of the former havens. Benacre Broad, Easton Broad, and The Meare at Thorpeness are still present today. Water from these broads reached the sea either by percolation through the shingle barriers, or by way of artificial cuts and sluices. At Minsmere, a sluice was initially constructed on the northern side of the estuary, near to the present Coney Hill (TM 478 670). The low lying land around Sizewell Belts may have at one time drained directly to the sea at Sizewell Gap, perhaps prior to the fourteenth century storms, although there is no documentary evidence for this. However, by 1783 the area drained northwards, east of the old chapel, into the broad at Minsmere. No map or geological evidence has been found to indicate that the Minsmere River ever drained southwards to enter the sea near Sizewell, as suggested by Pethick (1997, 2004b).

Frequent freshwater flooding at the end of the eighteenth and beginning of the nineteenth centuries prompted an Act of Parliament in 1810 to drain the former estuary, then known as Minsmere Level. Between 1810 and 1830, a new sluice was created further to the south, in approximately the same location as the estuary mouth in 1737. A new drainage ditch was also created, initially called Huntingsfields Ditch (on Bryant's map of 1826, Figure 12d), and later the New Cut (on the First Edition Ordnance Survey map of 1837, Figure 12e). A defensive bank also built between the sluice and the Minsmere end of the cliffs, at a height of *c.* 3 m above the surrounding marsh. This bank made use of a high area of land at Coney Hill, which itself was *c.* 3 m above the surrounding marsh. A second bank, running westwards from Coney Hill to the high ground north of Sheepwash Lane, was completed later between 1883 and 1903. The old river, with its sluice near Coney Hill, lost its identity between 1826 and 1837, and the Broad was reduced in size by further drainage, losing its identity by 1890.

Comparison of the maps in 1783 and 1837 demonstrates significant changes in coastline orientation to the north of Minsmere. In 1783, the coastline between Dunwich and Minsmere projected some 150 metres into the wider bay between Southwold and Thorpeness. The northern and southern ends of this promontory coincided with the limits of the Dunwich and Minsmere cliffs. In the following 50 years, the cliffs were apparently eroded very rapidly, the coastline moving *c.* 100 metres landwards. The coastline fronting Minsmere and Sizewell also retreated landwards, albeit at a slower rate than the cliffs, so that by 1837 the promontory had been eroded away completely, and a new slight promontory had been formed in the vicinity of Coney Hill. Further south, the coastline had retreated landwards to align with the northern limit of Sizewell Cliffs at Sizewell Gap.

Changes in the position of the coastline since 1837 have been illustrated by digitising the positions of mean high water (MHW), mean low water (MLW) and the coastline (generally highest astronomical tide, HAT) position from six editions of Ordnance Survey maps dating from 1837 to the present day (Table 1), using GIS software. This method of analysing change from historical maps has been used in several previous studies (e.g. Kenneth Pye Associates Ltd., 2004a and 2004b). Maps were initially geo-referenced using latitude and longitude scales in the map margins, which were converted to British National Grid (OSGB36) coordinates. Maps were then calibrated with a precision of *c.* 1 metre using six fixed points along the coastline: Cliff House, near Dunwich (TM 477 688), Coastguard Cottages, on Dunwich Heath (TM 477 678), the old chapel (TM 473 660), Minsmere sluice (TM 477 661), Sizewell Cottages (TM 475 628), and Coastguard Cottages at Sizewell (TM 475 627). It is estimated that this process of geo-referencing and calibration allows features on the 1:63,360 map of 1837 to be digitised with an accuracy of ± 8 m, and features on the other 1:10,560 maps used to be digitised with an accuracy of ± 4 m. The historical positions of MHW, MLW and the coastline have been superimposed on the most recent 1:10,000 Raster image supplied in digital format by the Ordnance Survey, and are shown in Figure 13.

The history of the coastline since 1837 can be summarised in two phases. Between 1837 and 1903, the Minsmere Cliffs retreated rapidly by *c.* 156 m, at an average rate of *c.* 2.3 m yr^{-1} (Table 5), until the southern end of the cliffs aligned north-south with Coney Hill. The Minsmere frontage to the north of the sluice also retreated landwards, but at a much slower rate of *c.* 1.1 m yr^{-1}. The Minsmere frontage to the south of the sluice, however, experienced

significant accretion between 1837 and 1903, moving *c.* 83 m seawards. The accretion occurred mostly in the first 50 years, an average rate of *c.* 1.7 m yr^{-1}. The historical position of the coastline in 1837 is represented on the ground by a secondary dune ridge, lying behind the present line of dunes. Between these areas of erosion and accretion, the sluice acted as an anchor point to the coast, with no change in the position of the coastline. The sluice has, however, acted as a fulcrum for a change in coastline orientation, which has rotated in an anti-clockwise direction.

Between 1903 and 1976 (the most recent survey of the coastline by the Ordnance Survey), the rate of coastal change reduced significantly. Very little change occurred to the coastline south of the sluice, the position of the coastline being stabilised by the presence of a dune ridge by 1903. The coastline between the sluice and Coney Hill has experienced some accretion since 1903, moving *c.* 20 m seawards. This has included creation of a new dune ridge, which provides additional flood protection during storm events. Minsmere and Dunwich Cliffs continued to erode, although at a reducing average rate of *c.* 1.3 m yr^{-1} in the 50 years between 1903 and 1953, and at *c.* 0.6 m yr^{-1} in the 50 years between 1953 and 2003. As the Minsmere and Dunwich cliff line retreated westward, so the coastline between Coney Hill and the end of the cliffs has also retreated, the dune ridge migrating landwards by 20 to 30 m.

3.2 Evidence from aerial photographs (1940 to 2003)

Six aerial photographic surveys of the coastline, dating from 1940, 1952, 1968, 1991, 2000 and 2003, have been obtained from several sources (Table 6). Although originally taken at different scales and covering different areas, these photographic surveys are reproduced in this report at an enlarged scale of 1:2500 (Figure 14). The survey of 2000 had been auto-rectified, allowing the historical coastline positions between 1937 and 1976 to be overlaid into the photographs using GIS software. A composite image of the whole coastline from Dunwich to Thorpeness in 2003 is also shown in Figure 15.

The aerial photographs from July 1940 show evidence of erosion and flooding following a severe storm event, probably that in 1938. To the north of Coney Hill, much of the dune ridge had apparently been eroded or flattened, with vegetation being removed back to the Minsmere

defensive bank. At Coney Hill, the sea had also breached the bank, with a large washover fan some 150 m wide and deep spreading over the marsh (also noted by Steers, 1951). To the south of Coney Hill, although the dune ridge appears to have been eroded on its seaward side, the ridge was still fairly well vegetated, and does not appear to have been breached. To the south of the sluice, where the dunes were higher, erosion appears to have been localised, with the coastline fronting what is now Sizewell Power station showing no sign of erosion.

In common with other flat and strategically vulnerable sections of coastline in south-east England, the Minsmere Levels were flooded in June 1940 as a Second World War defensive measure. The aerial photographs from July that year clearly show the flooded areas behind the coast. In addition to the flooding, concrete blocks were placed along the backshore (these are visible on the air photographs to the present day and act as reference points) and other anti-invasion hardware were installed on the beach (Figure 16). The sea sluice gate remained open until the end of hostilities in 1945.

The aerial photographs from 1952 indicate that the coast to the south of Coney Hill had recovered well from the 1938 storm, the dune ridge now being well vegetated. A ground photograph taken in 1950 (Figure 16), shows that the dune ridge was still relatively low at this time. However, the beach appears sandy, which would have aided accretion of the dunes. To the north of Coney Hill the coastline does not appear to have recovered well. There is little evidence of a dune ridge reforming, the artificial bank acting as the only effective defence. The washover fan at Coney Hill appears unvegetated and to have been recently active, possibly during the storm surge of 1949.

Although photographic evidence is not available, it is likely that the storm of 31 January to 1 February 1953 caused additional erosion to the coastline north of Coney Hill, reactivating the washover fan and removing any newly formed dunes. Following the 1953 storm, the backshore north of Coney Hill, in front of the artificial bank, was artificially planted, to encourage the formation of a dune ridge and thereby increase protection to the marsh behind (Figure 17). By 1968, aerial photographs show that a vegetated ridge had developed to the north of Coney Hill, while south of Coney Hill a new ridge, seaward of the original, was beginning to form. Evidence of the washover was beginning to disappear, although the narrowest part of the dune ridge still occurred at this point. Evidence from ground

photographs (Figure 18) suggests that by the 1970s the beach was becoming coarser, with shingle accumulating on the upper beach.

Aerial photographs from 1991 show continuing widening of the dune ridges, and increasing vegetation cover. To the south of Coney Hill, the new dune ridge has also widened. The line of concrete blocks, which in places formed the back of the beach in 1940, are now set well back from the beach. Very little change is apparent to the south of the sluice, except for the engineering works relating to the building of Sizewell B Power Station.

The photographs dating from 2000, overlaid with the coastline positions obtained from historical maps, together with the most recent photographs of 2003, provide a valuable insight into changes along this coastline over the last 160 years. To the north of Coney Hill, the coastline has progressively migrated landwards, linked to the receding Minsmere Cliffs. Since the last cartographic survey in 1976 (black line), the dune ridge has apparently moved 5 metres further inland. Between Coney Hill and Minsmere sluice, widespread erosion up to 1903 gave way to a period of dune accretion until the position of the coastline had stabilised by 1976, since which time very little change has occurred. To the south of the sluice, the 1837 coastline position is marked by the presence of an old ridge line. Following a period of accretion up to 1903, the position of the coastline stabilised, and apart from a short stretch of erosion some 250 m south of the sluice, the position of the coastline has remained virtually unchanged since 1976.

Ground photographs from December 2004 indicate that although the position of the coastline is now relatively stable, the composition of the beach is becoming coarser, with a high percentage of shingle on the upper beach (Figure 19). The lack of large quantities of sand will inevitably reduce the potential for dune growth. It is possible, however, that quantities of sand will return to the beach during lower energy wave conditions during the summer months. The overall tendency of the Minsmere shore to become more shingle-dominated over the past 50 years is probably related to decreased input of sand to the beach/nearshore system, associated with slower erosion of the Dunwich and Minsmere cliffs.

3.3 Evidence from bathymetric charts (1868 to 2004)

Approximately 20 historical Admiralty Charts have been examined in this study, dating from 1868 to 2004, in order to assess changes in the bathymetry between Southwold and Thorpeness. Although new editions of Admiralty Charts are published almost annually, charts include data from many surveys conducted at different times, sometimes decades apart. After an initial survey in 1845, the sea bed was largely unsurveyed for the next 100 years, except for Sizewell Bank which was important for navigation purposes. The next major surveys were conducted around 1940 and 1960.

Six charts were selected for digitising, dating from 1868, 1908, 1940, 1960, 1974 and 1992 (Table 7). These charts represent the first charts to be published after major resurveys, and although each will contain data spanning a number of years, they provide the best available indication of the situation at the given dates. Although quoted datum levels and map units changed several time over this period, a close inspection of the charts showed that depths in areas of the chart which remained unsurveyed were unaffected by quoted changes in datum levels. For this reason, datum changes are assumed to have affected depth values by less than one foot (smallest unit on the charts). Values have therefore been quoted relative to chart datum at the time of the survey, and converted to metres. Figure 20 shows the digitised sea bed bathymetry from the six charts. Interpretation of sea bed changes is assisted by consideration of changes in the position of the –5 and –15 m CD contours (Figure 21) and cross sectional profiles (Figures 22 and 23).

The bathymetry of the frontage between Southwold and Thorpeness has largely been influenced over the past 150 years by the morphology of offshore banks, some 2 km from the shore.

In 1868, there were two distinctly separate banks, Dunwich Bank in the north, and Sizewell Bank in the south. Sizewell Bank was by far the largest and most important. The bank had a teardrop shape, the highest part being *c.* 3 m below chart datum, with a slender ridge running for 7 km to the north. The bank had an amplitude of *c.* 8 m above the surrounding sea bed, running parallel to the coast *c.* 2 km from the shore. Dunwich Bank, however, was much smaller in extent (1.5 km only). It had an amplitude of only 3 m above the surrounding sea

bed, with the highest point some 7 m below chart datum. The bank also lay *c.* 2 km from the shore, near Dunwich. Close to the shoreline, the 1868 chart shows ridges and runnels running parallel to the coast some 300 metres from the shore. The runnels had a depth of *c.* 2 m, and extended from Dunwich to Sizewell.

Analysis of changes between 1868 and 1940 is not possible due to the lack of survey information. This is illustrated by the 1908 chart, which shows only that the southern part of Sizewell Bank had accreted towards the coast, although it does not appear to have accreted vertically. Subsequent charts in 1918 and 1922 show that the southern part Sizewell Bank accreted towards the north and east. The area to the south of the bank, offshore from Thorpeness, also accreted, the bank rising to just 2 m below chart datum. The foreshore at Thorpeness had also experienced significant accretion.

Resurveying in the late 1930s provided the first update on changes to the north of Sizewell since 1868. Changes in the intervening period had been significant. By 1940, Dunwich Bank had merged with the northern part of Sizewell Bank, to produced one long bank running parallel to the coast from Dunwich to Thorpeness, *c.* 2 km from the shore. The bank was generally 4 to 5 m below chart datum, with an amplitude of 5 to 6 m above the surrounding sea bed. The highest part of the bank was still the part nearest to Sizewell, and this had migrated *c.* 700 m closer to the shore. The increase in height of the bank offshore from Dunwich will have inevitably reduced wave activity along this part of the coast, and may well be responsible for the reduction in cliff erosion rates.

The chart of 1960 still shows a continuous ridge of the merged Dunwich and Sizewell Banks extending from Dunwich to Thorpeness, although a low point in the ridge had been created just to the south of the sluice at Minsmere, and Dunwich and Sizewell Banks were once again separate, discrete banks. The ridge was also showing signs of erosion, with a narrower, more sinuous morphology. Sizewell Bank was still the highest part of the ridge, although its crest had been lowered to *c.* 4 m below chart datum. Dunwich Bank had also been lowered, and had migrated *c.* 800 m landwards between 1940 and 1960. Conversely, the area to the east of the banks had accreted, the –15 m contour moving *c.* 250 m offshore. Overall, the bank system appears to have been lowered, but had also spread out to cover a wider area.

By 1974, erosion had apparently continued, with a deepening of the low point between the two banks, south of Minsmere sluice. The crests of the banks were still 3 to 4 m below chart datum, and had not migrated landwards since 1960, although they had both narrowed. The sea bed landward of these banks along the Minsmere frontage had also deepened to *c.* 11 metres below chart datum. Deepening in this area may be a consequence of reduced cliff erosion, with less sediment being introduced to the coast and nearshore zone. To the east, continued accretion had caused the –15 m contour to migrate a further 500 m offshore.

Erosion of the banks apparently continued through the 1980s, as the chart of 1992 shows both Dunwich and Sizewell Banks shrinking further in size, although largely maintaining their crest height. The banks had also migrated a further 200 m inshore. By 1992 the 'trench' at Minsmere, landward of the banks, had infilled slightly, although depths in excess of 10 m are recorded less than 1 km offshore from the Minsmere Sluice. Little change has been recorded in charts published since 1992, only the frontage between Minsmere and Thorpeness having been resurveyed. Sizewell Bank appears to be maintaining its lateral and vertical extent. The deep area landwards of the banks has infilled slightly, although depths of up to 10 m are still recorded near Minsmere. At Sizewell, local changes in the beach and nearshore morphology have undoubtedly been affected in the last 30 to 40 years by construction of piers, water intakes and discharge pipes associated with the power station.

Using GIS software, it has been possible to calculate the changing volume of sediment across the Dunwich and Sizewell Banks from the historical charts (Table 8). The area over which volumes have been calculated is shown in Figure 12. Calculations between –5 and –10 m depth illustrate changes on the banks, and show that the greatest changes occurred up to 1960, when the banks accreted in volume by *c.* 18%. The banks then eroded over the following decades, although the net change between 1868 and 1992 has been accretion of *c.* 8%. Calculations to a depth of –30 m indicate that, overall, the greatest changes occurred between 1940 and 1960, when the area to the east of the banks accreted and the –15 m contour migrated offshore. In part this may reflect offshore movement of sediment during and following the major storm surges of 1938, 1949 and 1953. This offshore area subsequently lost sediment between 1974 and 1992, although the net change between 1868 and 1992 was accretion of 20.5 km^3 of sediment, or a 2.3% increase.

Assuming an average recession rate of 1.49 m yr^{-1} for the Minsmere and Dunwich Cliffs, and an average cliff height of 10 m and cliff length of 3 km, it can be estimated that between 1868 and 1992 the cliffs supplied *c.* 5.5 km^3 of sediment to the nearshore zone. Over this same period, the coast to the north of Coney Hill also eroded, supplying an additional *c.* 0.2 km^3 of sediment, while the coast to the south of Minsmere accreted, amounting to *c.* 0.7 km^3 of sediment. It is therefore evident that erosion of the Dunwich and Minsmere Cliffs has contributed at most only one quarter of the increase in sediment volume of 20.5 km^3 on the Dunwich and Sizewell Banks and the area further offshore to the east. Based on a consideration of coastal and deeper marine processes, it is likely that the majority of sediment must be derived from further north along the coast, including the cliffs Easton, Covehithe or Pakefield, or from offshore.

3.4 Evidence from beach profile and bathymetric profile data (1991 to 2003)

Since 1991 the Anglian Region of the Environment Agency have surveyed beach profiles at a number of fixed transects as part of its Strategic Monitoring Programme. Surveys are carried out annually to low water, in both winter and summer. In 1992, 1997 and 2003, bathymetric surveys extended the beach profiles offshore to a distance of about 3 km (Figure 24). Data for these surveys have been examined for 8 profiles lines (shown in Figure 25), and sediment volume changes have been calculated (Table 9).

Data for the two most northerly profiles S1C6 and S1C7 indicate that the trend of lowering and landward migration of Dunwich Bank, identified from historical charts, has continued. Between 1992 and 2003 Dunwich Bank dropped *c.* 1 m in elevation, and migrated landward by 100 to 200 m. The area to the west of the bank, however, increased in elevation by *c.* 1 m, infilling with sediment. Profiles S1B1 to S1B5 to the south, each show a landward migration of Dunwich / Sizewell Bank by 100 to 200 m, and an infilling of sediment by 0.5 to 1.0 m to the west of the banks. There was, however, no net change in the level of the bank crest. Changes on the southernmost profile S1B6 show that the position of Sizewell Bank was stable between 1992 and 2003, with no net change in the level of the crest of the bank, although there was significant accretion in the area landward of the bank.

At all profiles, changes to the beach down to lowest astronomical tide (-1.6 m OD), were relatively small. All profiles show some reduction in beach volumes between 1992 and 2003, except profile S1B3, immediately to the south of the sluice, which has accreted slightly. However, the fluctuation in beach volumes from year to year is greater than the net erosion or accretion taken over the 11 year period. Considering the whole profile, down to –12 m OD, calculations show that all profiles accreted between 1992 and 2003, by between 7 and 10% for all profiles except S1B1, to the north of Coney Hill, which accreted by less than 5%.

3.5 Evidence from lidar data (1999)

Data from a lidar survey flown by the Environment Agency in March 1999 have been analysed using the methodology described by Blott and Pye (2004) to derive a digital elevation model (DEM) of the coastline between the Minsmere Cliffs and Sizewell, including the Minsmere Reserve. The DEM of the site is shown in Figure 26, and enlarged areas of the coastline to the north of Sizewell Power Station and the area to the north of Coney Hill are shown in Figures 27 and 28. A number of topographic cross-sections (locations on Figure 26) derived from the lidar DEM are shown in Figures 29 and 30.

Ground survey data provided by the RSPB were used to estimate the vertical accuracy of the lidar data by comparison of recorded elevations at the same grid references (Table 10). This analysis has shown that the lidar survey generally underestimates the height of the banks and sea walls by *c.* 0.3 to 0.6 m, since the horizontal resolution of the lidar data (2 metres) is similar to the width of the top of the banks. Lidar spot heights therefore frequently represent an average elevastion of the crest and sides of the banks. Conversely, the lidar survey generally overestimates the height of the flat, low-lying ground between the banks and sea walls due to reflections from the upper part of the vegetation canopy. The overestimation is greatest for the reedbeds, where the vegetation is tallest, the lidar data being *c.* 0.5 to 0.7 m higher than the ground survey data. Differences between the techniques are smaller for the grassland areas, where the lidar data is generally *c.* 0.1 to 0.3 m higher than the ground survey data.

The lidar DEM illustrates that large areas of land behind the coastal defences are very low lying, and vulnerable to flooding in the event of a breach. Much of the land across Minsmere Level and Leiston Marsh, to 3 km inland, lies between 0.6 and 0.7 m OD. Areas to the north of the Minsmere Old River and Coney Hill are slightly higher, at *c.* 1.5 m OD. These areas drain onto Minsmere Level by gravity flow.

The standard of defence provided by the dune ridge can be interpreted using the lidar DEM, and a long-shore parallel profile derived from it along the crest of the dune ridge (Figure 30). The ridge fronting Sizewell Power Station is generally high (mostly in excess of 5 m OD). There is one low section, between northings 264000 and 264300, where the dune ridge drops to less than 5 m OD, corresponding to the end of the engineering works for power station construction. However, the land behind this section is protected by an artificial earth bank built 200 m behind the dune ridge, and by the relict dune ridge corresponding to the 1837 coastline position. Construction of an additional barrier between the earth bank and the relict dune ridge is planned by the power station to remove any remaining threat of marine flooding in this area.

Between the power station and Minsmere sluice, the dune ridge is wide and high (in excess of 6 m OD), and for much of this length is backed by the earlier dune ridge. For 700 m north of the sluice, to northing 266800, the dune ridge is narrower and lower, with a crest height between 4.5 and 5.0 m OD. The dunes along this section are, however, fairly wide and well vegetated. The ridge between this point and Coney Hill then becomes lower, at between 4.0 and 4.5 m OD, although this section is still defended by the second ridge a few metres behind. North of Coney Hill the dune ridge is the lowest and narrowest on the whole frontage. In places the ridge crest lies below 4 m OD, including the location where a small breach occurred during the storm of 14 to 15 December 2003. This 200 m section is the most vulnerable stretch of coastline between Dunwich and Sizewell, and represents the most likely location of a major breach occurring during a future storm surge.

4. Summary of coastal morphology changes and relationship to processes and sediment supply

Significant changes have taken place along the coastline between Dunwich and Sizewell in the last 700 years. The former Minsmere Estuary, whose marshes were so prone to coastal flooding that by 1360 Leiston Abbey had to be relocated 3 km inland, had by 1780 been isolated from the sea by the development of a shingle barrier. Several other estuaries along the Suffolk coast became blocked around this time. Historical map evidence has shown that up to 1737 the Minsmere River had its entrance near to the present sluice, and then later flowed further to the north with its entrance near to Coney Hill.

Pethick (2004b) suggested that around this time the river had its entrance much further to the south, immediately to the north of the Sizewell B Power Station, with a shingle ridge extending northwards to the Dunwich cliffs, forming a spit feature. However, there is no map evidence that the Minsmere River ever flowed this far south, or that a large shingle spit ever existed on this section of coast. Pethick bases his hypothesis, at least in part, on relict dune ridges identified from a composite lidar image which appears to show a recurve at the southern end, hypothesised to reflect proximity to a neighbouring tidal channel. However, detailed inspection of the original lidar data and aerial photographs in this study has shown that the apparent recurves are actually areas of vegetation rather than sedimentary/ morphological structures. Analysis of historical maps has shown that the relict dune ridge extending north from the Sizewell B Power Station represents the position of the coastline in *c.* 1837. There is also no evidence from historical Admiralty charts of an ebb tidal delta along this part of the coast, as hypothesized by Pethick (2004b).

In the early nineteenth century, recession rates of the Dunwich and Minsmere Cliffs were very high, at *c.* 2.3 m yr^{-1}. This seems to correspond with a period of higher southerly sediment drift associated with more northerly and northeasterly winds. The Dunwich Bank also did not extend this far north at this time, and there would have been less storm wave 'protection' than at present. Were it not for the presence of the Minsmere sluice, which has acted as an anchor point since its construction, the whole shoreline to the south of the cliffs would probably have moved inland in sympathy with the receding cliffs. Since the coastline was essentially fixed at the sluice, only the northern Minsmere shoreline retreated, leaving a slight promontory at the

'bedrock' high of Coney Hill. The sediment from the eroding cliffs was transported southwards with the prevailing drift, where it accumulated to the south of the sluice. Over the following 80 years, the whole frontage at Sizewell accreted seawards by some 80m. Some sediment was probably also transported further to the south.

Pethick (2004b) suggested that the Minsmere sluice has acted as a headland, with a shallow bay (termed 'Sizewell Bay') forming to the south. Historical map evidence obtained in this study report suggests the opposite is the case. Sizewell Bay was present before the construction of the sluice, and the sluice has in fact aided accretion of the Sizewell frontage.

By the early twentieth century, cliff erosion at Dunwich had reduced to *c.* 1.3 m yr^{-1}, possibly as a consequence of more westerly winds, and due to the greater sheltering effect of Dunwich and Sizewell Banks which had enlarged, probably due to the previous input of sediment from the cliffs. The coastline to the south of the sluice outfall had also stabilised, and has remained relatively stable to the present day.

By the mid twentieth century, the cliff recession rate had reduced further to *c.* 0.6 m yr^{-1}, and following the storm surges of 1938 and 1953, the frontage of Minsmere has remained relatively stable. The sluice appears to be acting as a sediment groyne, permitting some accumulation of sediment on its northern side. In this section, between the sluice and Coney Hill, there has been some accretion where a new foredune ridge has developed. In the immediate shadow zone to the south of the sluice, a reduction in sediment supply has allowed some localised cliffing of the seaward side of the dunes during storms, although the coastline between the sluice and Sizewell has remained stable since 1903.

The only section which appears to be actively eroding, albeit slowly, is the area to the north of Coney Hill. Pethick (2004b) has suggested that the change in coastline orientation since the nineteenth century has been caused by a reducing sediment supply causing a change from a drift-aligned to a swash-aligned coast. It is more likely that erosion at the northern end of the Minsmere frontage is in response to the landward retreat of the Minsmere cliff line. Reduced cliff erosion in recent years has undoubtedly reduced sediment supply to the shore north of Coney Hill. Winnowing of the finer gravel and sand fractions has led to a loss of beach volume and relative coarsening in this area. Wind-blown sand transport from the beach has

virtually ceased in this area, and consequently the back beach 'ridge' between Coney Hill and the end of the Minsmere Cliffs has a low elevation; it is therefore the most likely location for overtopping during storms.

Recent erosion of the frontal dunes between the northern end of the power station site and the Minsmere Sluice is a specific local phenomenon which is probably related to the development of a relatively deeper water channel between Sizewell Bank and Dunwich Bank. Especially during periods of easterly winds larger waves are able to pass through this gap in the banks and reach the shoreline. Average water levels in this area may also be raised slightly as a result of wind-driven 'set-up'. However, the dunes are relatively wide and/or high in this area and there is little risk of large-scale flooding or a major breakthrough in the foreseeable future.

5. Options for future management

In June 2004 the Environment Agency published a draft consultation document outlining the options for future flood risk management on the Minsmere to Sizewell coastline (Environment Agency, 2004b). A revised version of the document will be issued in due course. The main options for consideration identified in the draft document are essentially the same as for other frontages under consideration in Suffolk and Norfolk:

(1) Do Nothing or No Active Intervention
(2) Do Minimum
(3) Hold The Line:
    a) Beach recharge
    b) Offshore breakwaters
    c) Rock armour groynes
(4) Managed Realignment
    a) Frontage realignment
    b) Mid realignment
    c) Extensive realignment

Preliminary comments on each of the main options are made below.

5.1 Do Nothing or No Active Intervention

This option would involve the ceasing of all maintenance, repair and renewal work to the sluice, and there would be no active management or repair to embankments or dune system. As a result of this option, continued erosion to the northern part of the Minsmere frontage would be expected. Overtopping and/or breaching of the dune ridge during high tides would not be repaired, which would result in further lowering of the dune ridge at these locations during subsequent high tides. Ultimately, erosion of the embankment behind the dune ridge could lead to breaching of this second line of defence, causing tidal inundation of the Minsmere Reserve. In the absence of a severe storm surge of 1953 proportions, this is likely to be a relatively slow and progressive process, taking perhaps 30-50 years, even allowing for

projected sea level rise. However, there is a significant risk that a major storm surge, which could occur at any time, would breach both the outer ridge and the secondary clay bank defence line north of Coney Hill, creating a washover type feature similar to that seen on air photographs of the 1940 to 1954 period. This would cause damage to the freshwater habitats in the northeastern corner of the Reserve but in itself would not necessarily cause serious damage throughout the Reserve.

5.2 Do Minimum

This option allows for repair of the defences if damage occurs, and minor preventative works, but does not provide for major routine maintenance of the defences or improvement of the standard of defence.

The implications of such a strategy for the Minsmere frontage are that any minor works undertaken are unlikely to have any significant impact on the direction of average medium term rate of coastal change. The Coney Hill to Minsmere cliffs frontage will continue to be vulnerable, as under the 'Do Nothing' approach.

5.3 Hold The Line

Hold The Line options involve maintaining or improving the standard of coastal defence to hold the existing defence line at its current position. Three options for holding the line have been identified for consideration, namely beach recharge, offshore breakwaters, and rock armour groynes. These options could be implemented in conjunction with other measures, such as a partial realignment of defences for part of the site, or changes to the tidal sluice (these will be considered in the next section).

The Hold The Line option is unlikely to be sustainable for the Minsmere Reserve frontage in isolation, since recession of the cliffs to the north is certain to continue, and this will place erosion pressure on the northern two-thirds of the Reserve shoreline. Any scheme relating to

Minsmere would have to involve the entire coastal zone extending at least from Southwold to Thorpeness.

Hold The Line – Beach Recharge

This option utilises addition of sediment to the beach and/or dunes to replenish sediment lost through erosion. By increasing the beach volume, the beach would become wider and flatter, and increase the dissipation of wave energy, thereby improving the standard of defence. It is likely that some fine sand would be transported to the dune ridge, increasing its size and stability, further improving the standard of defence. A proportion of the sediment would also be transported southwards along the coast, effectively recharging these areas as well. Ultimately, only a small amount of sediment is likely to be lost entirely from the system, since studies have shown that sandy sediment arriving at Thorpeness will be transported to the offshore banks, with a potential for increased dissipation of wave energy (although it is unlikely that the volume of sediment reaching the banks as a result of a beach recharge project would be too small to have any significant impact on wave energy).

Pethick (2000a), in his response to the Environment Agency Consultation Document (Environment Agency, 2004b) states that beach recharge at Minsmere would retard the natural evolution of the bay by forcing the beach plan-form to re-orientate anticlockwise, increasing erosion through longshore drift. At present there is little scientific evidence to support such a hypothesis. While longshore drift will undoubtedly increase to some extent following beach recharge, there is little basis on which to expect the planform of the 'bay' to change radically. Increased beach width and height along the shore would create more effective wave energy dissipation. While nourishment with shingle would offer some advantages in terms of energy dissipation and reduced longshore sediment losses, nourishment with sand or mixed sand/ shingle would offer greater potential for dune ridge construction at the back of the beach.

Experience from previous beach recharge projects has shown that regular recharges are required, usually on an annual basis, to replace the inevitable loss of the initial recharge material. Also, the particle size of the recharge material is crucial for the scheme to be a

success, dependent on project objectives and prevailing coastal process constraints. A thorough study would be required to evaluate the feasibility, costs and environmental implications of such a scheme.

Hold The Line – Offshore Breakwaters

This option would involve the construction of offshore breakwaters parallel to the shore along the Minsmere Cliff to Sizewell frontage, to reduce wave energy and net sediment erosion rates. Breakwaters could have a major impact on sediment transport processes along this coastline, and would not allow for any natural adjustments to the alignment of the coast in responses to changes further north. Rock breakwaters would have the added detrimental effect of a substantial visual intrusion to an area of outstanding natural beauty. It is also highly unlikely that the use of rock breakwaters could be justified on economic or safety grounds unless there was significant increase in the threat posed to the Sizewell nuclear facility.

Hold The Line – Rock Armour Groynes

This option would involve the construction of rock groynes perpendicular to the shore along the Minsmere Cliff to Sizewell frontage, to reduce longshore drift rates and reduce erosion. These structures would potentially have an even larger impact on coastal sediment processes and aesthetic environmental quality than offshore breakwaters. Any scheme at Dunwich or Minsmere would undoubtedly have a major negative impact further down drift at Sizewell in the medium term.

5.4 Managed Realignment

Managed realignment would require identification of a preferred landward line of defence and a plan for the (perhaps staged) retreat of the shoreline. Three possible options for realignment have so far been identified by the Environment Agency. All would involve the loss of part or all of the Minsmere Reserve in its present form. Any loss of habitat due to realignment of sea

defences would require compensation in the form of replacement habitat elsewhere. Even if it were possible to recreate comparable habitat elsewhere, this would involve a considerable economic cost to the Agency, and could indirectly affect the local economy of the area through loss of visitors.

Managed Realignment – Frontage Realignment

This option would involve relocating the present first line of defence *c.* 150 m inland along the whole coastline between Minsmere Cliffs and Sizewell. This defence line would encroach into the Minsmere Reserve with some loss of freshwater habitats. This option would allow the coastline at the northern end of the Minsmere frontage to retreat in sympathy with the cliffs.

The key to the success of such a scheme would be the position of the Minsmere sluice. In the Environment Agency plan, the position of the outfall would be maintained although the implementation of an additional sluice or replacement with a pumping station would be considered. Analysis earlier in this report has shown that the sluice has acted as an anchor point to the coastline. A frontage realignment would therefore require a landward movement of the sluice to maintain the alignment with the coastline to the north. This would also require the landward movement of the outfall, since in its present position it would act as a sediment groyne restricting longshore drift along the newly aligned coast. Although this scheme would benefit the coastline to the north of the sluice, by realigning the shore with the cliffs, the coast to the south of the sluice could experience erosion. Historical evidence has shown that accretion along the part of the coast was only possible due to the present location of the sluice. Removal or 'set-back' of this part of the shore-dune system would also have a potentially serious effect on the Sizewell defences.

It is also important to consider how such a realignment would be implemented. While some areas could be allowed to retreat, or 'roll-back' to the new defence line naturally, the sluice and outfall would require active setback. In the short-term this could result in substantial erosion and southward transport of sediment before the defences could roll-back to the new line. In this way, a large volume of sediment in the dune ridges could be lost rather than acting as a new defence.

Managed Realignment – Mid or Extensive Realignment

These two options would both involve relocating the line of defence to the higher ground which surrounds the Minsmere Reserve, either as far as a line between Sandypytle Plantation and South Belt Woods (up to 1.4 km inland), or as far as Dam Bridge at Eastbridge (up to 2.6 km inland). Flood barriers would be required at the landward limit to protect assets upstream.

Both these options would result in the loss of many of the current habitats on the Minsmere Reserve, including the fresh and brackish water pools, freshwater reedbed, and grazing marsh habitats. There would also be a substantial economic cost due to the loss of these habitats (in terms of compensation). These options would also be very expensive, requiring the construction of flood barriers and sluices to protect areas to the west and south of the new tidal area, including Sizewell which is vulnerable due to the low lying land of the Sizewell Belts.

There is little scientific evidence at present to suggest that there is a need at the present time for such a large scale realignment of the defences along much of the coast. Apart from some local cliffing of the dune ridge, the coast to the south of Coney Hill has remained relatively stable for the past 50 years, and over a longer timescale the coastline south of the sluice has accreted substantially.

Pethick (2004a) expresses the opinion that maintenance of the existing habitat at Minsmere is not a sustainable option, and the long-term (> 100 years) outlook for the Minsmere wetland must involve its conversion from fresh water to saline water habitats. Pethick (2004a) also suggests that there is a need for a short to medium term strategy to warp up the Minsmere wetland, by permitting inundations through the sluice, as a precursor to a full scale transition to tidal habitat. This is based on the view that the surface elevation of the wetland is now 1 to 2 m lower than MHW, and that in 100 years there will be a shortage of fine sediment availability which may prevent development of inter-tidal habitat. Analysis of cross-sections across the Minsmere Reserve from lidar data (Figure 29) has shown that the elevation of the marsh is generally between 0.6 and 0.7 m OD, dropping to 0.4 m in some places. Tidal levels at Minsmere (Table 1) are 0.8 m OD at MHWS and 0.4 m OD at MHWN (MHW is therefore *c.* 0.6 m OD). While there may be some errors in both the lidar and tidal data (perhaps up to

0.2 m), it is clear that the marsh is not substantially lower than MHW, and certainly not 1 to 2 m lower as stated by Pethick (2004a). It is therefore difficult to justify such a strategy of inundations through the sluice, particularly considering the economic and habitat implications of permitting saltwater flooding of the reserve.

There is also considerable uncertainty surrounding the impact of restoring the Minsmere tidal 'estuary'. Pethick argues that reopening or removing the tidal sluice would lead to the reformation of an ebb tidal delta which would reduce water depths and provide a wave dissipater for the Minsmere to Sizewell shoreline. At present there is no scientific evidence to support such a hypothesis.

The southern limit of the current large-scale realignment provisionally suggested by the Environment Agency would occur *c.* 1 km north of the Sizewell Power Station. The high ground at this point is Goose Hill, *c.* 300 m behind the current coastline. To the south, the low lying land continues to the Sizewell Belts. Any realignment scheme would therefore require a defensive barrier to protect against flooding of the Sizewell Belts.

6. Conclusions and recommendations

There is no evidence that rates of coastal recession and/or frequency of flooding on the Dunwich – Minsmere – Sizewell frontage have accelerated in recent decades, as a result of sea level rise, increased storminess or any other factor. Indeed, the evidence indicates that the opposite has occurred. This situation could change in the future if predictions about increases in the rate of sea level rise and storminess prove to be correct. However, the effects of such changes might not be significant for at least 30 to 50 years. At present, there is no scientific basis to conclude that a major coastal realignment, involving virtual abandonment of the freshwater habitat interests at Minsmere, is required in the short to medium term.

Rates of cliff erosion at Dunwich, and shoreline recession at the northern end of the Minsmere Reserve have actually declined in the past 50 years. Reduced sediment input to the nearshore system in this area has resulted in a loss of beach volume and a change in beach sediment character between Minsmere Cliffs ad Coney Hill. The beach and back-beach ridge in this area are currently narrow and relatively low; consequently minor overtopping can be expected on an annual basis and there is a significant risk of major over-topping and breaching of the secondary line of defence during a big storm event.

Retreat of the northern Minsmere frontage has been retarded over the past century and a half by the New Cut sluice, which acts as an 'anchor point'. Removal of this hard point can be expected to result in a relatively rapid landward adjustment of the shoreline both the north and immediately south of the sluice.

At the present time there is healthy accumulation of shingle (and also sand on the lower beach) in the vicinity of the sluice, and no direct local requirement to reposition the sluice landward.

There is, however, a requirement to allow some natural readjustment of shoreline position between Minsmere Cliffs and the former position of Coney Hill. In order to allow the future cliff erosion, the secondary defence line in this area should be moved back, perhaps by as much as 100 m, creating an energy dissipation zone between the bank and the active beach ridge within which natural washovers can be accommodated.

An alternative would be to strengthen the west-east aligned bank (North Wall) from Coney Hill to the higher ground inland, and to accept that washover and increased saline influence will become inevitable in the northeastern corner of the Reserve. Such a strategy would require changes to the drainage system within the Reserve.

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Project: Bats and noise in impact assessment

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Abstract

Whilst many studies have been published in the last few years which consider the potential effects of noise on bats, none of these contains sufficient information about sound levels to enable a reliable threshold for disturbance to be derived. However, based on the research which is available, it is possible to assess a level below which one can be confident that noise disturbance is unlikely for bats in roosts, when commuting and during foraging. Further research is called for with input from both an acoustics specialist as well as a bat specialist being critical to obtaining usable data.

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A Review of Disturbance Distances in Selected Bird Species

M. Ruddock & D.P. Whitfield

A report from Natural Research (Projects) Ltd to Scottish Natural Heritage

2007

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Summary

Animals' avoidance of humans or human activities can have several adverse effects on their distribution and abundance, and a frequent tool used by conservation managers to avoid such effects is to designate 'buffer zones' (or set-back distances or protection zones) around centres of animals' distribution within which human activity is restricted.

A common method used to prescribe buffer zones involves one or two measures of disturbance distance: 'alert distance' (AD), the distance between the disturbance source and the animal at the point where the animal changes its behaviour in response to the approaching disturbance source, and 'flight initiation distance' (FID), the point at which the animal flushes or otherwise moves away from the approaching disturbance source.

Recommendations on 'safe-working distances' (essentially, buffer zones around breeding sites) have been made for a number of UK breeding bird species, but without any objective justification. With recent changes in Scottish legislation on human access to the countryside and protection of some breeding birds' nest sites there was therefore a need to review available information on disturbance distances for 26 'priority' bird species which breed in Scotland.

Preliminary assessment revealed few previous studies quantifying disturbance distances for the study species, and so an expert opinion survey was conducted in which opinion was solicited on 'static' and 'active' disturbance distances (i.e. AD and FID, respectively) when birds were approached by a single pedestrian when incubating eggs and when with chicks. The survey resulted in 89 respondents providing 1083 opinions on disturbance distances.

It was difficult to validate independently the results of the survey, because relatively few empirical studies had been conducted on disturbance distances for the study species. Subjectively, however, the survey appeared to give similar

results to those of research based on quantified field observations, although distance estimates in the expert survey may have been slightly high in some species.

A number of descriptive statistics for AD and FID are presented for each of the species from the expert opinion survey, including upper distances which incorporated 90 % of opinions on AD (although it is highlighted that AD is probably impossible to measure in practice for many species when breeding). Species accounts, describing the results of a literature review for each study species (and related species) on disturbance distances, are also presented and include published information on AD and FID, responses to a number of disturbance sources, previously prescribed buffer zones and forestry practices, when relevant.

Expert opinion is typically used as a stopgap in research as a bridge between empirical evidence and policy (although our review suggested that it is probably frequently misused in this field by not being a temporary measure and with insufficient validation) and, given the shortage of empirical field studies, it is recommended that the expert survey results should be regarded as preliminary until further validation has been undertaken. It is suggested that such validation should include further analyses of the survey in relation to predictions of AD and FID in the literature, and more field studies of disturbance distances. Encouragement for observers to measure disturbance distances in national monitoring schemes of breeding birds is recommended as being especially useful as a mechanism to generate empirical data on disturbance distances rapidly.

1. Introduction

Animals commonly move away from an approaching human or encroaching human activities such as recreation and this response can have adverse influences on, for instance, their feeding success (Burger & Gochfeld 1998, Fernández-Juricic & Tellería 2000), range use (Andersen et al. 1997), reproduction (Giese 1996, Miller et al. 1998), survival (Wauters et al. 1997, West et al. 2002) and abundance (Miller et al. 1998, Fernández-Juricic 2000, 2002). Human disturbance is increasingly becoming a concern to conservationists because as human populations continue to expand, ecotourism is increasing as a potential revenue source, and wildlife in diminishing areas of refuges are exposed to greater human recreational and other anthropogenic activities (Wight 2002, Christ et al. 2003).

While predicting the effects of humans on wildlife is difficult (Knight & Cole 1995, Hill et al. 1997, Carney & Sydeman 1999, Gill et al. 2001, West et al. 2002) one of the most frequently exploited tools used by land managers and policy-makers when promoting co-existence of wildlife and people is the creation of 'buffer zones' (or set-back distances or protective/management zones) around potentially sensitive centres of wildlife activity (e.g. nest sites of rare, protected or uncommon bird species, or breeding colonies) within which human activity is, at least in principle, restricted or excluded with the objective of minimizing disturbance impacts (Holmes et al. 1993, Knight & Temple 1995, Rodgers & Smith 1995, 1997, Richardson & Miller 1997).

Two broad steps have been used to prescribe buffer zones (Knight & Temple 1995, Richardson & Miller 1997, Fernández-Juricic et al. 2005). In the first step the distance at which humans should be separated from wildlife (minimum approaching distance) is estimated, and then the areas where humans should not encroach to avoid displacing wildlife (buffer zones) are prescribed. Several methods have been proposed or employed to calculate minimum approaching

distance (MAD) and buffer zones (e.g. Anthony et al. 1995, Rodgers & Smith 1995). The most common method used to estimate MAD is to observe the reactions of subject animals to the approach of a single disturbance source, typically a pedestrian. One or two metrics are recorded: alert distance (AD), the distance between the disturbance source and the animal at the point where the animal changes its behaviour in response to the approaching disturbance source (specifically, in birds, when the head is raised in an alert posture: Fernández-Juricic & Schroeder 2003), and/or flight initiation distance (FID), the point at which the animal flushes or otherwise moves away from the approaching disturbance source.

This method can be criticized for several reasons: 1) animals may react at greater distances to grouped disturbance sources (e.g. a group of pedestrians) (Beale & Monaghan 2004a, Geist et al. 2005) or react differently to different disturbance sources (Rodgers & Smith 1997, Stalmaster & Kaiser 1997); 2) reaction distances may be less in birds which are less capable of withstanding the effects of disturbance (e.g. those more stressed by low food availability or poor body condition: Gill et al. 2001, Beale & Monaghan 2004b); 3) the availability of alternative habitat may affect tolerance of disturbance and hence FID and AD (Gill et al. 2001, West et al. 2002); 4) direct approaches may elicit greater FID than tangential approaches (Burger & Gochfeld 1981) although the reverse may also occur (Fernández-Juricic et al. 2005); 5) many other factors may affect FID or AD, including animal group size (Burger & Gochfeld 1991), stage of breeding (Bauwens & Thoen 1981), prior exposure to disturbance and/or habituation to disturbance (Burger & Gochfeld 1991, Ruggles 1994), exposure to human persecution or hunting (Ferrer et al. 1990, Louis & Le Berre 2000, Galeotti et al. 2000) or, even, observers' clothing colour (Gutzwiller & Marcum 1997).

Understanding most of these potential influences on FID and AD is improved by considering that animals perceive humans as potential predators and react

accordingly (Frid & Dill 2002, Beale & Monaghan 2004a) and hence individuals should vary FID and AD dynamically so as to minimize the costs of disturbance whilst maximizing the probability of survival and/or reproduction (Ydenberg & Dill 1986, Lima & Dill 1990). While individual behavioural indicators such as FID may not reflect population impacts (Gill et al. 2001), use of this theoretical framework, the risk-disturbance approach (Frid & Dill 2002), can facilitate the development of strategies for co-existence of wildlife and people (Beale & Monaghan 2004a, Blumstein et al. 2005), and in many management situations behavioural indicators may be essential (Blumstein et al. 2005), notably those involving nest sites when a centre of animal activity through a fixed location can be safely assumed.

The application of observed behavioural indicators of disturbance distance to the designation of buffer zones has seen a wide range of methods, several of which are problematic in that they are unlikely to satisfy the practical objective of minimizing disturbance impacts on wildlife. For example, application of average measures of FID to prescribe buffer zones is unlikely to prevent all birds from being disturbed (Götmark et al. 1989, Fernández-Juricic et al. 2005) and most studies of behavioural reactions to a disturbance source record only FID, but buffer zones based directly on FID does not allow for any adaptation of wildlife to occur before disturbance affects animals' presence, and so AD probably has greater utility (Rodgers & Smith 1997, Fernández-Juricic et al. 2001, 2005).

Rigorous detailed studies such as that of Fernández-Juricic et al. (2005) can undoubtedly contribute towards the development of scientifically-defensible applications in practice, but a more fundamental issue is that designated buffer zones often have no obvious empirical basis in behavioural studies on the relevant species. For example, several European countries have legislative protective buffer zones for white-tailed eagle *Haliaeetus albicilla* nest sites (Helander & Sjernberg 2003), but there are no published estimates of FID and AD in this species (B. Helander pers. comm.). Indeed, while the scientific

literature on human disturbance is vast, surprisingly little is devoted to empirical measures of FID and/or AD (see Results). Body mass and FID are positively related in birds (Blumstein et al. 2005) and Blumstein et al. (2003) have shown that FID may be a species-specific trait. Such analyses may therefore provide potential mechanisms to extrapolate buffer zones from existing measures of disturbance responses which may relieve some concern over the shortage of species- or site-specific measures of FID or AD.

In Scotland, recent separate legislation has provided for greater freedom of peoples' access to the countryside and increased protection of the nest sites of birds from reckless disturbance (see also Beale & Monaghan 2004a). Provisional guidance has been produced on 'safe working distances' (effectively, recommended disturbance-free zones) around the nest sites of several bird species (Currie & Elliot 1997), but the source of these recommendations was not given and no scientific justification was apparent. Thus, there was a pressing need for such justification to be gathered for many bird species; however, our preliminary investigations revealed a shortage of published empirical disturbance studies. Gathering novel data on FID and AD on many breeding species was practically impossible, but recalling that many scientists and experienced fieldworkers have previously routinely visited the nests of all species in the course of research projects, bird ringing and other monitoring programmes, our approach was to survey expert opinion for data on these measures. While seldom explicitly acknowledged, expert opinion often appears to be used in the designation or recommendation of buffer zones (e.g. Grier et al. 1993a, b, Petty 1998) and is frequently used as a method to bridge the gap between research evidence and practical policy or procedure implementation in other fields (notably medicine). Expert opinion is potentially a very powerful tool because information can be rapidly and cost-effectively gathered, especially in the present context when researchers or other fieldworkers necessarily (although cumulatively) disturb birds at breeding sites frequently but such disturbance events and the behavioural response of the disturbed birds are not routinely documented.

Here, we present the results of an expert opinion survey on FID and AD estimates for several species of birds when breeding and assess the method's robustness by comparison of these estimates with published empirically derived estimates. We also present the results of a literature review of available information on disturbance for the selected study species and close relatives.

2. Methods

Our survey covered 26 bird species considered as a priority by Scottish Natural Heritage, the government's statutory advisor on nature conservation in Scotland, largely based on breeding species which are either listed on Annex 1 of European Union (EU) Wild Birds Directive (79/409/EEC) or are otherwise rare in Scotland, and substantially following the list considered by Currie & Elliot (1997). To avoid undue repetition the full species list is given later (Results: Table 1): due to sample size and close ecological similarity, two species, common crossbill *Loxia curvirostra* and Scottish crossbill *L. scotica* were considered together. For two lekking gamebird species, capercaillie *Tetrao urogallus* and black grouse *T. tetrix*, disturbance of both parental females and lekking males was considered.

Expert opinion was solicited from three main sources: authors of published literature on the survey species when breeding, members of Scottish Raptor Study Groups (SRSGs; fieldworkers with considerable experience in monitoring breeding raptors: see http://www.scottishraptorgroups.org/ and Hardey et al. 2006), ringing (banding) groups and British Trust for Ornithology (BTO) Nest Record Scheme recorders. Selected experts were asked to complete a questionnaire form which requested that they record the distance at which individuals of the species for which they had experience typically showed a 'static' and an 'active' behavioural response to a single pedestrian observer walking in full view towards an active nest or bird(s) with chicks. 'Static' disturbance distance was defined as the distance at which there was a static

behavioural response to the disturbance stimulus (= observer), such as increased vigilance and/or alarm calling (i.e. AD). 'Active' disturbance distance was defined as the distance at which there was an active behavioural response to the disturbance stimulus (= observer), for instance taking flight, moving away from/towards the observer (i.e. FID). Potential respondents were asked to record separately the typical disturbance distances for incubating birds and for birds with chicks.

Hence, for each species, opinions on four distances were solicited, with the exception of capercaillie and black grouse when opinion was also garnered on AD and FID for lekking birds. By way of acknowledgement that the survey could not be precise, potential respondents were asked to record their opinion on each disturbance distance in one of 10 categories (m): <10, 10 – 50, 50 – 100, 100 – 150, 150 – 300, 300 – 500, 500 – 750, 750 – 1000, 1000 – 1500, or 1500 – 2000.

Opinion was solicited preferentially via email, supplemented by postal and telephone requests. A total of 503 emails and letters were sent of which 69 emails failed, due to non-existence of address, and 89 generated responses with a completed questionnaire form i.e. a 20.5 % response rate, providing 1083 opinions of disturbance distance. A small number of respondents replied without completing a questionnaire, usually because they felt that typical disturbance distances could not be quantified given intraspecific variation.

We did not ask potential respondents to document the extent of their experience with the survey species, to avoid discouraging responses through requesting too much information initially. Rather we organised an ad hoc request to a random sample of 10 positive respondents with a background in ringing and raptor monitoring, and a sample of six BTO Nest Scheme Recorders. On average each of the 16 respondents visited 8.8 ± 1.9 SE nests per species per annum and had 16.4 ± 1.5 SE years of experience per species.

We calculated descriptive statistics for each species' disturbance distances (mean and, as data were not normally distributed, median) and for distances with a sample of opinions ≥ 10 we calculated the "80 % range" which involved the range of distances without the most extreme upper 10% and lower 10 % of opinions e.g. for a disturbance distance with a sample of 20 opinions the highest two and lowest two opinions were excluded. (For species with < 10 opinions we were forced to take the lower and upper recorded limits as the "80 % range".) The resultant upper value was thus the point at which 90 % of respondents considered that disturbance would have occurred. This metric was calculated due to its potential equivalence to 90 % of the cumulative probability of observed AD or FID; 90% or 95% values are frequently reported in observational disturbance studies (e.g. Holmes et al. 1993, González et al. 2006). The establishment of protective buffer zones based on 90 – 95 % of FID cumulative frequency distribution has also been considered an effective strategy in protecting nesting raptors (Olendorff & Stoddart 1974, Suter & Joness 1981, Mersmann & Fraser 1990) and has been examined in detail by Fernández-Juricic et al. (2005). Calculation of survey values on an equivalent 'upper percentile' metric thus gave us greater scope to compare the survey results with those from observational behavioural studies and designated or recommended buffer zones in the published literature. We estimated 90 % AD rather than 90 % FID because as noted by Fernández-Juricic et al. (2005), among other authors, AD represents a better metric with regard to potentially preventing any form of disturbance than FID, even though some studies have found AD difficult to record (e.g. González et al. 2006).

We conducted a literature search through ISI Web of Knowledge, Web of Science, Google Scholar, Google and other contemporary search engines, using the key-word 'human disturbance' for papers and reports on disturbance distances and buffer zones in the study species and allied species. For each paper or report reviewed we extracted available measures of buffer zones and FID or AD descriptive statistics.

3. Results

Expert opinion survey

A summary of the survey results are given in Table 1 and the full results are presented in Appendix 1. In two species (*Aquila chrysaetos* and *Haliaaetus albicilla*) some opinions on AD were at such large distances (≥ 1 km) that we considered it unrealistic that an observer could have noted routinely the behavioural change required under AD criteria and, as these extreme opinions clearly contradicted the majority opinion, these extremities were ignored. As a further caveat, for several species the number of opinions on disturbance distances were low, and so these results should be viewed with caution.

Published literature on AD and FID

Full details of the literature review for each of the study species are given in Appendix 1. For our 26 study species we could find comparable published data from observational studies of only six species, for FID only, and for only one species (*Asio otus*) were disturbance distances recorded for both incubating and chick-rearing birds (Table 2). This prevented us making any formal statistical comparison of the survey results with the published AD and FID literature. There were nevertheless indications of consistency between published measures of observed disturbance distances and surveyed expert opinion in most of the six species, although there were some instances when the opinion survey suggested higher disturbance distances than revealed by empirical observations (Appendix 1). Similarly, there also appeared to be broad support of the expert opinion survey from other sources of information on human disturbance in the literature, although again this assessment was necessarily somewhat subjective (Appendix 1). An incidental finding of this review for the 26 study species and related

species was that data on FID were more frequently available from North American studies than European studies.

4. Discussion

Buffer zones are one of the most common tools used by conservation managers when attempting to promote co-existence between wildlife and humans yet surprisingly we found that it was relatively infrequent that they were based on empirical measures of disturbance distances, especially in Europe. In part and, we suspect for North American cases in particular, this may have been because such measures had been documented but were unavailable to us. It was also evident that in some cases buffer zones had probably been designated on the basis of disturbance distance research on another population of the same species: such transference may be justified if disturbance distances are relatively constant and species-specific (Blumstein et al. 2003, 2005) but there are several examples where this does not appear to hold (see white-tailed eagle account, Appendix 1). Although many methods have been used to adapt measures of AD or FID to describe MAD or buffer zones (Fernández-Juricic et al. 2005), in other cases it was apparent that alternative methods, other than measuring AD and/or FID, had been used to prescribe MAD. For example, distances between white-tailed eagle nests and human habitation, and analyses of breeding success in relation to distance to human habitation have been used to designate buffer zones for this species (Helander & Stjernberg 2003, Helander et al. 2003). It was also apparent, nevertheless, that in many cases expert opinion had been used in recommending and designating buffer zones, although rarely acknowledged explicitly. Hence, we suggest that the main method employed by our study, expert opinion, is actually much more prevalent in this research field than would be superficially apparent from the literature and is probably more common than use of observed AD or FID. The prevalence of expert opinion in the field of buffer zone designation is arguably at odds with the method's more typical temporary role in policy formulation as a stopgap for research evidence, and perhaps

indicates that, generically, expert opinion may have usurped the role of empirically-derived field studies in this important conservation field. Hence, it would appear to be imperative that more observations of AD and FID on many species should be collected.

Some caution should perhaps be exercised when comparing the expert survey results with 'safe working distances' recommended by Currie & Elliot (1997), which was the main previous review with comparable disturbance distances for most of our study species. First, Currie & Elliot's (1997) review involved recommendations for forestry workers, which includes forms of disturbance not covered by our study. Second, Currie & Elliot (1997) included disturbance to nest-building birds; a stage in the breeding cycle which we did not include in the expert survey. It was apparent, nevertheless, that in most species the expert survey yielded disturbance distances which were lower than the recommendations of Currie & Elliot (1997) and, in sharp contrast, indicated that distances were greater when birds were with chicks than when incubating. Currie & Elliot (1997) do not indicate the source of their recommendations, although given the shortage of empirical information for the species which were reviewed some form of expert opinion was probably involved. It also seems likely that Currie & Elliot (1997) included a sensitivity criterion based on birds' changing propensity to abandon a breeding attempt at different stages of the breeding cycle in response to disturbance; this would explain why 'safe working distances' were greatest for nest-building birds and lowest for birds with chicks. Both Currie & Elliot (1997) and the present study probably primarily involved expert opinion, but as the present study included an attempt at validation and was explicit in its sources and methods, we suggest that the present study may form a more objective basis for assessment of MAD and, potentially, buffer zones.

The substantial variation in our expert opinion survey for most species was interesting in that it reflected the finding in many observational studies of substantial variation in disturbance distances between individual birds and

circumstance (Appendix 1). It was clear from the literature review, and from the expert survey, that there are considerable differences in the distances at which birds of the same species respond to disturbance and this suggests that whenever possible buffer zones should be responsive to such differences. Several factors likely underlie this variation (see Introduction) but one major source of such differences is the type of disturbance and how it relates to the form of disturbance to which birds may already be exposed (e.g. Stalmaster & Kaiser 1997, Rutz et al. 2006). Therefore, since the expert survey was based on opinion for behavioural responses to a single approaching pedestrian, application of the results to other disturbance sources may be inappropriate. Other forms of disturbance may invoke reaction at greater (e.g. pedestrian group with a dog) or lower (e.g. motor car) distances. The variation in birds' responses to disturbance presents a major challenge when designating buffer zones and, as pointed out by Fernández-Juricic et al. (2005), argues for the process to involve great care, rigour and sensitivity to the factors which may underlie this variation.

AD has been recommended as the most appropriate measure on which to base buffer zones (Rodgers & Smith 1997, Fernández-Juricic et al. 2001, 2005) but in several of our study species it is difficult or impossible to measure, either because of the distances involved or because birds on nests are hidden from view (see also González et al. 2006). This probably explains why respondents less frequently provided opinions on 'static' disturbance distances for several species and must temper interpretation of the expert survey results. This will obviously be a widespread difficulty for nesting birds, because typically birds conceal their nests, and is an issue which has not been considered by those studies that have recommended the use of AD in buffer zone designation which often involved perched or foraging birds (e.g. Fernández-Juricic et al. 2001, 2005). If AD tends to be a fixed proportion of FID, as has been recently suggested (Cárdenas et al. 2005, Gulbransen et al. 2006), then this may offer scope for the use of an AD surrogate when only FID measures are available. Since there was also a shortage of empirical measures of FID and, especially,

AD for the majority of our study species against which to validate the opinion survey we therefore would urge that the results should be regarded as preliminary estimates until further validation assessments are undertaken (and in line with how expert opinion should usually be treated - as a research stopgap). We should also emphasise that in some cases disturbance distances according to expert opinion were apparently slightly higher than comparable empirical observations and so the lower limits of the distance categories we used should probably be preferred.

Further work to validate the expert opinion survey could include a comparison of 'observed' survey results for AD against expectations based on empirical studies of other species (Blumstein et al. 2005) and, of course, further field studies of disturbance distances in the study species. On the latter requirement, at least in the UK, and probably in many other countries, numerous breeding attempts are visited annually as part of national monitoring schemes (such as the BTO Nest Record Scheme in UK) and incorporation of a protocol for observers to measure disturbance distances during visits would be relatively straightforward but yield a large volume of information from across a wide geographical area. Field studies dedicated to particular species which are poorly served by reports to national monitoring schemes would also probably be required, although it should not be necessary to undertake such field studies on all of the species which are viewed as a priority.

5. Acknowledgements

This study would not have been possible without the generous cooperation of a large number of experts who freely gave of their time and experience: we are extremely grateful to all the survey respondents for their assistance and patience. We also thank the BTO for putting us in touch with contributors to the Nest Record Scheme.

Marsh harrier *Circus aeruginosus*

Previous studies

Aside from 'natural' factors which may influence breeding marsh harriers, such as flooding of nests (Crivella et al. 1995), prey abundance (Simmons 1994, Dijkstra & Ziljstra 1997, Fritz et al. 2000), agricultural cultivation schemes, and predators (Underhill-Day 1984, Dijkstra & Ziljstra 1997, Stanevicius 2004), the wetland habitat requirements of marsh harriers (e.g. Baldi & Kisbenedek 1998) often predisposes them to co-existence with wildfowl shooting interests. The detrimental effects of this association with human activity and the negative effects on marsh harriers are well documented particularly the consequences of lead shot ingestion from prey species (Pain et al. 1993, Pain et al. 1997, Mateo et al. 1999), which varies according to the stage of the shooting season (Pain et al. 1997, Mateo et al. 1999). In Britain, 8.7% of nest failures have historically been attributed to human disturbance (Underhill-Day 1984) and the energetic costs and detrimental effects of disturbance to this species have been rigorously examined (Fernandez & Azkona 1993). Nest attendance, food deliveries per hour and defence behaviour were all increased when a male marsh harrier was widowed (Fernandez & Azkona 1994). Nest defence against conspecifics is greater in males (Fernandez & Azkona 1994) although no mention is made of any gender-related differences in aggression towards human intruders.

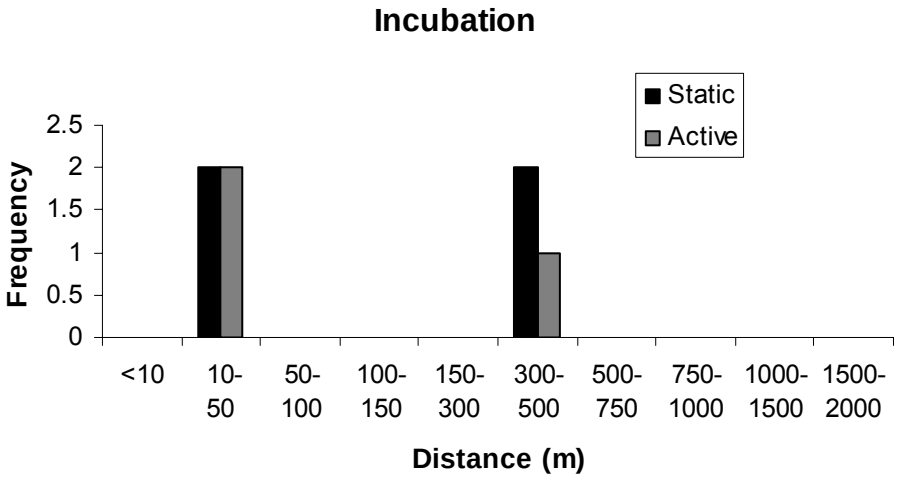
The effects of human disturbance can limit marsh harrier parental care in severe cases according to findings by Fernandez & Azkona (1993). In this study observations were made at a distance of 500 m from nest locations (duration of 549.5 hours), presumably the distance at which observers were not considered to affect harrier behaviour. Human activity around harrier nests was 5 - 10 intrusions per day on mid- week days and 50-100 intrusions on weekends with some nests more heavily disturbed than others. The majority of potential disturbance originated from crayfish trappers, which were captured within the

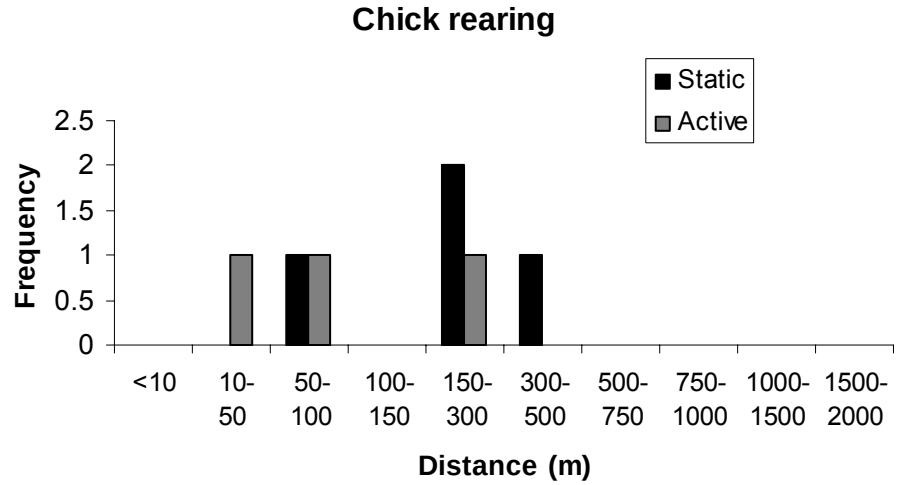
reedbed structure. Return times following flushing ranged from 1 - 89 min and was not dependent on the breeding stage. Nest attendance and incubation was less during high disturbance periods and was significantly different between sexes (♀>♂) and highest during incubation for females. Male behaviour was less affected by disturbance, since food provisioning by males was affected significantly reduced only in the incubation stage. Breeding success was unaffected, between disturbed and undisturbed pairs, suggesting harriers have developed coping mechanisms for the increased disturbance. However, total food provisioning of young, by both parents, was reduced by two-thirds in disturbed versus undisturbed nests and the nutritional status of chicks, measured by blood sampling to determine urea levels, revealed greater states of malnutrition in chicks from disturbed sites. Fernandez & Azkona (1993) also suggested that increased energy expenditure for adults was created by disturbance through increased vigour and requirement of defensive flight behaviour (high circling, alarming and stooping). The lack of observed reaction to crayfish trappers, visiting the nesting areas to check traps, was remarkable, suggesting habituation, and that birds were tolerant of regular sources of disturbance, but energetic and physiological costs to adults and chicks were likely to be higher in disturbed nest locations.

Marsh harriers appear to prefer reedbed nest site locations which are away from the land (presumably to minimise risks from land-based predators) and away from water-reedbed edges (presumably to avoid open water and reduce the risk of flooding); these locations also tend to reduce the risk of human pedestrian and boat disturbance (Stanevicius 2004). Nest site distance from open water in Stanevicius' study ranged from 15.1 - 46.9 m and 7.2 – 52 m from the shore: the author suggested birds were not actively disturbed until someone entered the reeds immediately beside the nest and boat-induced flushing from the nest was not observed during three years work on 55 breeding pairs. Detection of disturbance by marsh harriers is difficult due to habitat constraints imposed by

reeds that obscure the field of vision for both harriers and researchers alike (B. Riedstra, pers. comm.).

Expert survey results





Distance (m)	Incubation		Chick rearing	
	Static (4)	Active (3)	Static (4)	Active (3)
mean	215	153	231	110
median	215	30	225	75
"80%" range	10-500	10-500	50-500	10-300

Species summary

Currie & Elliott (1997) did not include marsh harrier in their recommendations and the survey opinion suggested 300 - 500 m may be an adequate buffer, although sample sizes were low for this species and a degree of caution may be necessary for static disturbance opinions given the difficulty that observers must have in seeing static cues of disturbance in this species. The upper value agrees with the distance at which Fernandez & Azkona (1993) made their behavioural observations. There is a degree of protection offered by the reedbed environment which reduces both the visible detection of disturbance by the birds and the likelihood of 'casual' human disturbance, but the studies on this species display the detrimental effects that disturbance can have on physiology and breeding performance. The reedbed nest site means this species is likely to flush most frequently at close range whilst on the nest, although as per hen harriers this does not mean they have not detected the disturbance source, and the survey opinion suggested that flushing may occur at greater distances, perhaps because extensive reedbeds are uncommon in the UK.

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Testing a novel agri-environment scheme based on the ecology of the target species, Montagu's Harrier *Circus pygargus*

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Farmland birds are in steep decline and agri-environment schemes (AES) to counteract these biodiversity losses are expensive and inefficient. Here we test a novel AES, 'Birdfields', designed using detailed ecological knowledge of the target species, Montagu's Harrier *Circus pygargus*. Current AES, such as field margins, that aim to improve foraging conditions (i.e. vole densities) for harriers are inefficient, as prey are difficult to capture in tall set-aside habitat. 'Birdfields' combines strips of set-aside to boost vole numbers and strips of alfalfa, as voles are accessible after alfalfa has been harvested. We found that vole numbers were generally highest in set-aside. Montagu's Harriers fitted with GPS-loggers used 'Birdfields' intensively after mowing, preferring mown to unmown strips. Thus, prey availability appeared more important than prey abundance. Thus, 'Birdfields', as a targeted AES for Montagu's Harriers, is more effective than previous AES due to increased prey accessibility. An additional advantage of 'Birdfields' is that it is considerably cheaper, due to the harvest of alfalfa. We advocate that AES should always include monitoring and research activities, aiming at a more adaptive conservation approach.

Keywords: agri-environment schemes, biodiversity, *Circus pygargus*, Common Vole, conservation, farmland birds, GPS-tracking.

In recent decades, farmland breeding birds have experienced dramatic population declines as a result of the intensification of agricultural practices (Donald *et al.* 2001, Guerrero *et al.* 2012). In Europe, agri-environment schemes (AES) are widely used to counteract biodiversity loss in agricultural ecosystems. However, their effectiveness is often poorly monitored or could not be shown (Kleijn *et al.* 2001, 2004, 2006, Bradbury & Allen 2003, Kleijn & Sutherland 2003). Positive effects are mainly found when AES are targeted to a specific species (Peach *et al.* 2001, Perkins *et al.* 2011, Pywell *et al.* 2012, but see Bright *et al.* 2015). In a world that needs to feed an ever-increasing human population, the high expenses and apparently low effectiveness make current AES unsustainable, calling for cheaper and more effective measures (Vickery *et al.* 2004, Baker *et al.* 2012).

A rare example of a scheme that has led to an increase of the target species' population concerns the Montagu's Harrier *Circus pygargus* in East Groningen, The Netherlands (Trierweiler 2010). This Red-listed, ground-breeding raptor almost became extinct in The Netherlands at the end of the 1980s (Zijlstra & Hustings 1992). However, a population of nearly 30 breeding pairs established in East Groningen when farmland was set aside on a large scale in the early 1990s (Koks *et al.* 2007). This population further increased after the introduction of AES in 1997 (Koks *et al.* 2007) to around 60 breeding pairs in 2011. In East Groningen,

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Montagu's Harriers rely greatly on the Common Vole *Microtus arvalis* and both laying date and clutch size are directly related to vole abundance (Koks *et al.* 2007). Moreover, annual population growth rate directly correlates with vole abundance (Koks *et al.* 2007).

The paradox of the successful Montagu's Harrier conservation is that although AES harbour higher vole densities (Koks *et al.* 2007), hunting Harriers capture most prey on intensive grasslands, where prey is less abundant but more accessible directly after mowing events (Trierweiler 2010). Hence, prey availability seems more important than prey abundance alone (e.g. Douglas *et al.* 2009). Based on these observations, we designed a novel AES, coined 'Birdfields', which combines both increased prey abundance and enhanced prey availability. 'Birdfields' consist of alternating strips of set-aside and alfalfa *Medicago sativa* (see Fig. 3 and S2). Set-aside consists of a mixture of cereals, grasses and herbs (Wiersma *et al.* 2014, Table S1), and its most important function is to enhance local densities of voles. Alfalfa is harvested three times per year, and its main function is to enhance prey availability (i.e. during and directly after mowing). An additional advantage of alfalfa is that it reduces the overall costs of 'Birdfields', making this a more economical alternative to current AES (see Discussion).

Here we evaluate 'Birdfields' by describing the ecological determinants of Montagu's Harrier response to this novel AES. This pilot study provides an example of how detailed knowledge of the ecology of the target species has helped to design a more efficient AES for Montagu's Harriers, advocating that ecological research is fundamental to enhance the effectiveness of AES.

METHODS

'Birdfields'

Two 'Birdfields' were created in spring 2011 in the Vriescheloërvennen, East Groningen, The Netherlands, close to a core breeding area of the Montagu's Harrier, one at Polderweg (16 ha, 53.09°N, 7.12°E) and another at Bisschopsweg (20 ha, 53.09°N, 7.11°E, Fig. S1). Half of each field was sown with set-aside and the other half with alternating strips of alfalfa and set-aside (Fig. S2). Birdfields were monitored in subsequent seasons (2012–2013) after the vegetation had fully developed. Alfalfa and, within the context of this particular pilot study, half of the set-aside were mown twice during the Harrier breeding season (in 2012 Polderweg was mown on 22 June and 30 July, Bisschopsweg on 12 June and 30 July; in 2013 both fields were mown on 10 June and 25 July). After each mowing, all edges of mown and unmown parts were georeferenced and digitized maps were created in ArcGIS 10.1 (Fig. S2).

Vole abundance

The relative abundance of voles in different crops was estimated by counting all vole burrows within 1 m of a 100-m transect line (Franken 2012). Six transects were counted per field, with two transects placed in the middle and four transects within 10 m of the edges of the field. Average field size was 12.4 ha (4.4–17.6, 1st–3rd quantiles, $n = 160$) in 2012 and 13.9 ha (6.3–18.0, $n = 79$) in 2013. We did not distinguish between active and inactive vole burrows as a pilot study revealed that even burrows we deemed inactive were used by voles. Counting vole burrows is known to be a relatively inaccurate method (Delattre *et al.* 1990) but it is the only practical method that allows monitoring voles throughout the large home-ranges of Montagu's Harriers. Counts were performed in all common crops in the study area (winter cereals, summer cereals (2012 only), alfalfa, extensive grassland, intensive grassland, rapeseed) and in AES (field margins, set-aside fields, 'Birdfields'). In total, 184 and 89 fields were counted in 2012 and 2013, respectively (Fig. S3).

The relative abundance of voles in set-aside and alfalfa was estimated directly after each mowing event by counting all signs of voles (vole burrows, runways between burrows and small food collection places consisting of pieces of herbs and grasses) in 1-m^2 plots. Plots were counted every 20 m, aligned in the centre of mown strips. In total, 2335 and 3849 plots were counted in 2012 and 2013, respectively. To gain insight on the nature and diversity of potential prey species, all small animals killed or disturbed by the mowing machine were counted during each mowing event, by walking directly behind the mowing machine; 20.6 and 73.3 km were covered in 2012 and 2013, respectively.

Response of Montagu's Harriers

Since 2009, 30 Montagu's Harriers have been equipped with UvA-BiTS GPS-loggers (Bouten

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et al. 2013) for ongoing studies of home-range size and habitat selection (Klaassen *et al.* 2014). Birds were trapped close to the nest using a mist-net baited with a stuffed raptor. GPS-loggers were attached using a 6-mm Teflon harness. In 2012, six male Montagu's Harriers equipped with GPS-loggers were breeding within 10 km of the 'Birdfields', of which one's nest failed in the incubation phase. Of the remaining five, four males regularly visited the 'Birdfields' (individuals ID#677, ID#669, ID#704, ID#505). The nest of ID#677 was predated during the incubation period. In 2013, only four birds fitted with a GPS-logger bred within 10 km of the 'Birdfields', three returning birds and one other individual (ID#669, ID#704, ID#505, ID#582). All four visited the 'Birdfields'. The distance between the nests and 'Birdfields' ranged from 2.7 to 5.6 km.

The frequency with which GPS locations are collected can be changed remotely with UvA-BiTS. GPS-loggers normally collected GPS positions every 5 min during the day and every hour during the night. At mowing events, GPS-loggers collected GPS positions at 3-s intervals ('high resolution measurements') when entering the area of the 'Birdfields'.

For analysis of the general use of 'Birdfields', high-resolution data were subsampled to a 5-min interval. Subsequently, daytime hunting positions were selected, assuming that all positions where the bird was flying were related to hunting. Although these hunting positions include some commuting flights, the proportion of time spent commuting is less than 5%, as commuting flights generally are relatively fast (A.E. Schlaich, R.H.G. Klaassen unpubl. data). The instantaneous speed obtained along with each GPS registration (Bouten *et al.* 2013) was used to distinguish between flying and sitting, using a threshold of 1.2 m/s (minimum of a two-peaked frequency distribution of speed values). Subsequently we calculated, per day, the percentage of hunting positions on 'Birdfields' relative to the total number of daily hunting positions. To evaluate whether the intensity of the use of 'Birdfields' was affected by mowing, the daily proportion of the use of 'Birdfields' was averaged over three periods: (1) 3 days before mowing, (2) the day of mowing and 2 days thereafter, and (3) 3–5 days after mowing. For this analysis, data for all four mowing events and the two 'Birdfields' were combined.

For analysis of habitat selection of Harriers within 'Birdfields', all GPS data, including high-resolution data in period (2) (i.e. 72 h after start of mowing, cf. above), were used. To evaluate whether Harriers preferred mown strips over unmown strips and whether they showed a preference for mown alfalfa or mown set-aside, the percentage of positions on mown alfalfa, mown set-aside and unmown set-aside was calculated and compared with the availability of these habitats, as calculated from the digital ArcGIS maps.

Statistical analyses

All analyses were performed in R 2.15.2 (R Core Team 2012). Model selection and validation followed recommendations in Zuur *et al.* (2009).

The mean number of vole burrows per transect in different crops was modelled with a generalized linear model using the R function *glm.nb* in package *MASS* version 7.3-35 (Venables & Ripley 2002). Using the mean of six transects per field, the number of vole burrows per field was explained by fixed effects *year* and *crop type* under a negative binomial distribution (count data with a small mean (12.7) and much larger variance (1059.3), thus with overdispersion). As the model including the interaction term *year* and *crop type* had a higher Akaike information criteria value (1641.7 vs. 1636.5), the interaction term was removed from the final model. A multi-comparison *post-hoc* test was performed to evaluate differences between the crops using R function *testInteractions* from package *phia* version 0.2 (De Rosario-Martinez 2013).

Vole abundance in the different 'Birdfield' habitats was modelled with a generalized linear mixed model using the R package *MCMCglmm* version 2.21 (Hadfield 2010). Because 57% of all observations were zeros, we used a zero-inflated model with a Poisson distribution. Signs of vole activity were modelled as a function of the fixed effects *year* and *habitat*, with *field* and *mowing event* as random effects. The model including the interaction between *year* and *habitat type* had a higher DIC-value (Hadfield 2010), and therefore the interaction term was removed from the final model.

The effect of mowing (periods *before*, *mowing* and *after*) was tested in a generalized linear mixed model using the R function *glmmPQL* in package *MASS*. The average percentage of 'Birdfield' use was modelled as a function of the fixed effect *period*, with *individual* and *year* as random effects with a binomial distribution.

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To test if Harriers preferred mown strips, a compositional analysis was conducted (Aebischer *et al.* 1993) using the R function *compana* in the *adehabitat* package version 1.8.15 (Calenge 2006). Proportions of surface area (based on digital habitat maps) were averaged per habitat category over the two mowing events and the two 'Birdfields' ('habitat availability'). Proportions of habitat use (based on the number of GPS positions in alfalfa, mown set-aside or unmown set-aside) were averaged per individual over the two mowing events and the two 'Birdfields' ('frequency of habitat use'). Subsequently, frequency of habitat use was compared with habitat availability in the compositional analysis including four birds and three habitat categories. Separate analyses were conducted for the 2 years.

RESULTS

At least five times more vole burrows were found in set-aside habitats compared with all other crops (Fig. 1a, Table S2, glmm: difference in deviance 311.97, df = 6, $P < 0.001$, for *post-hoc* test see Table S3). Also within 'Birdfields', there were more signs of vole activity in set-aside than in alfalfa strips (glmm: $P < 0.001$; Fig. 1b, Table S4). This emphasizes the importance of set-aside for high prey densities.

By walking behind the mowing machine, a rough impression about the nature and diversity of potential prey species in 'Birdfields' was obtained. In total, 197 small mammals were encountered, 176 Common Voles, six Common Shrews *Sorex araneus*, one Bank Vole *Myodes glareolus* and one

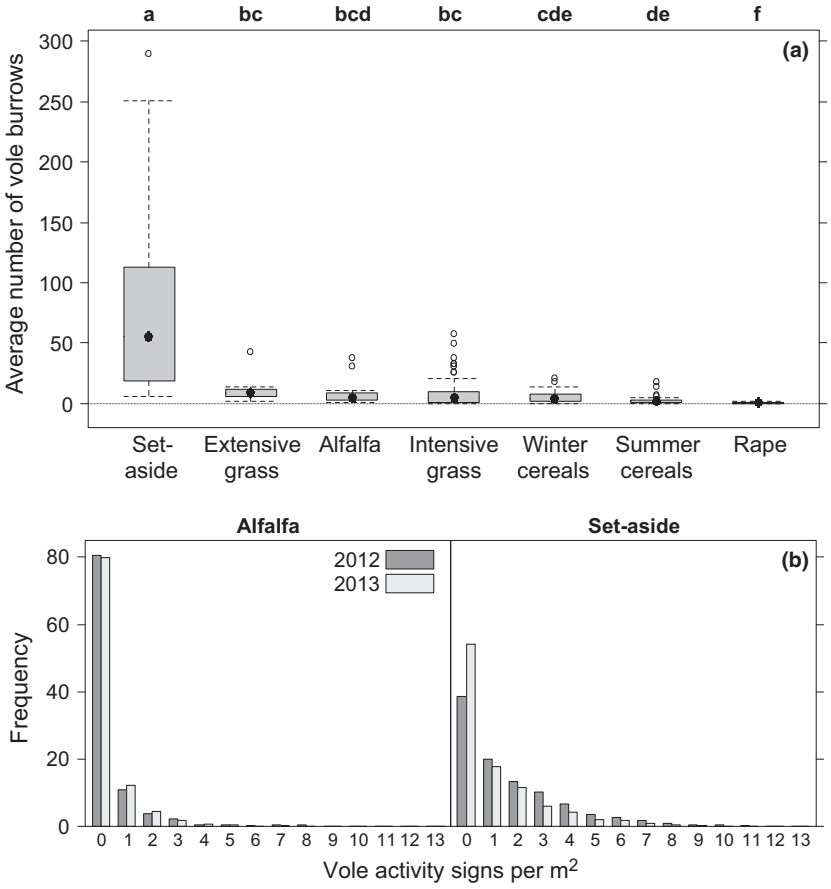


Figure 1. (a) Average number of vole burrows as counted along 100-m transects in different crops in 2012 and 2013 ($n = 272$ fields). Boxplots show median (dot), 25th and 75th quantiles (box), extremes (whiskers) and outliers (points). For sample sizes see Table S2. There was no significant difference between years (glm: difference in deviance 1.12, df = 1, $P = 0.29$). Crop types sharing letters did not differ significantly. (b) Frequency distribution of signs of activity of voles in 1-m^2 plots ($n = 6184$) on 'Birdfields' in 2012 and 2013, for alfalfa and set-aside. Less vole activity was recorded in 2013 than in 2012 (glmm: $P < 0.001$; Table S4).

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Wood Mouse *Apodemus sylvaticus*. Twelve mammals could not be identified. In all, 135 amphibians were recorded: 102 Common Toads *Bufo bufo* and 33 frogs *Rana* sp. Furthermore, two juvenile Common Whitethroats *Sylvia communis* and one juvenile Reed Bunting *Emberiza schoeniclus* were found. In 2013, 62 grasshoppers *Tettigonia viridissima* were encountered. Although amphibians were relatively abundant, they form a negligible fraction of the Harrier diet (Koks *et al.* 2007).

Montagu's Harriers fitted with GPS-tracking devices varied greatly in their general use of 'Birdfields' and this variation did not seem to be related to the distance to their nests (Fig. 2a, Fig. S4). For example, in 2012, of the three males (ID#669, ID#704, ID#505) breeding at approximately the same distance, only ID#505 used the 'Birdfields' intensively (Fig. 2a; ID#677 was a failed breeder). However, mowing had a strong effect on the intensity of the use of 'Birdfields' in all individuals. Harriers visited 'Birdfields' more during and after mowing than before (glmm: $t = 5.506$, df = 17, $P < 0.001$ and $t = 2.204$, df = 17, $P = 0.042$, respectively; Fig. 2b). In addition, Harriers selected mown set-aside over mown alfalfa, but primarily selected harvested habitats (alfalfa and harvested set-aside) over unmown habitat (unmown set-aside, compositional analysis: in 2012: $\lambda = 0.036$, df = 2, $P < 0.01$; in 2013: $\lambda = 0.075$, df = 2, $P < 0.01$; Figs 3 & 4).

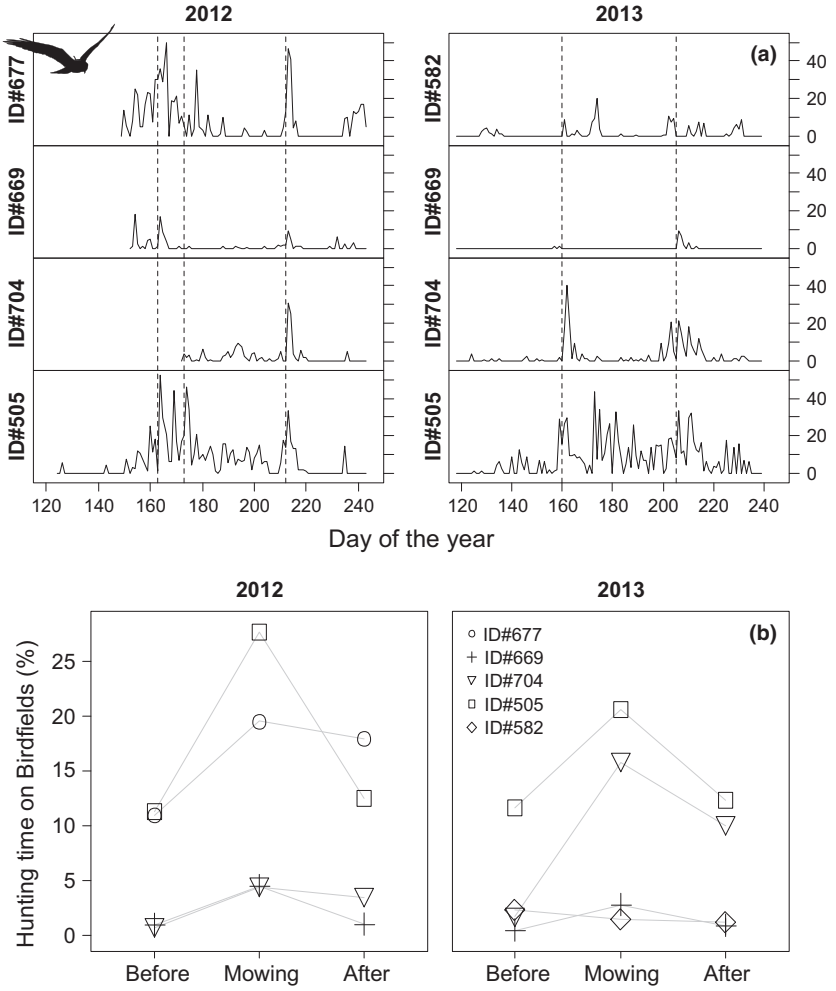


Figure 2. (a) Use of 'Birdfields' by four male Montagu's Harriers in 2012 and 2013 as percentage of their daily hunting time. Dashed lines indicate mowing events on 'Birdfields'. (b) Mean use of 'Birdfields' by male Montagu's Harriers as a percentage of their daily hunting time during the 3 days before mowing, day of mowing and 2 days thereafter, and 3–5 days after mowing.

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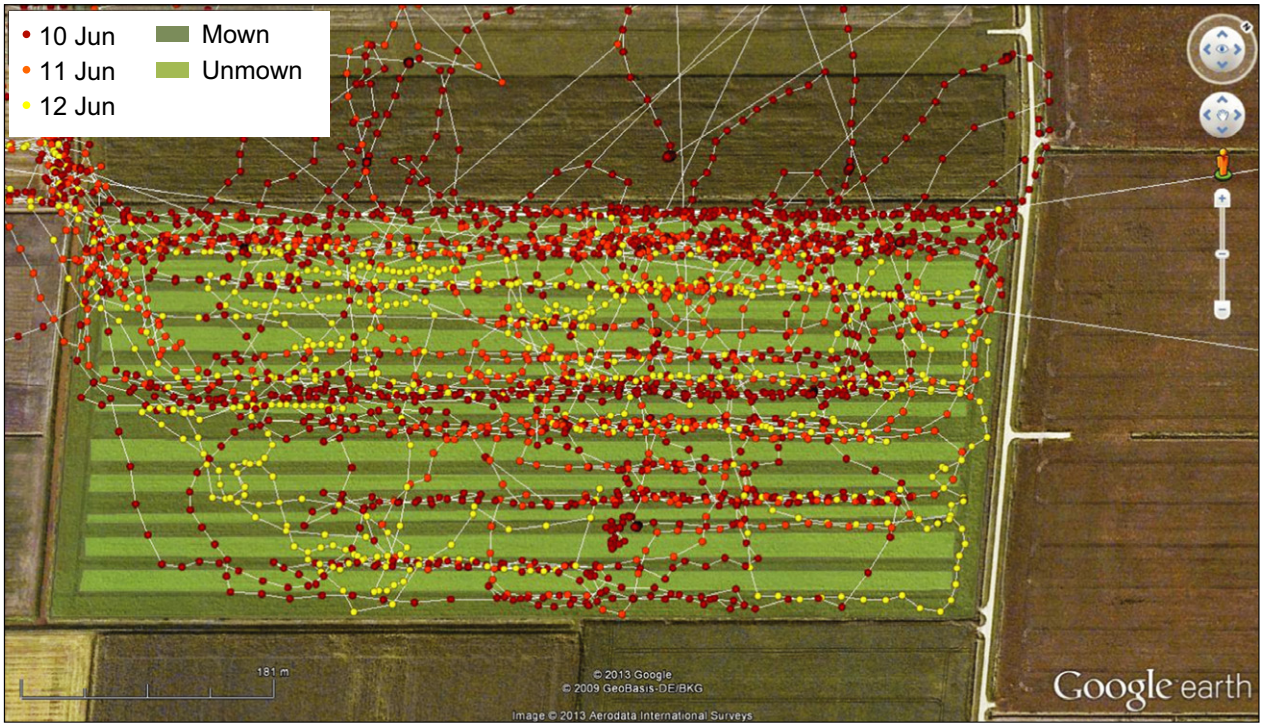


Figure 3. Example of the tracks of male Montagu's Harrier ID#704 hunting above a 'Birdfield' on the day of mowing (10 June 2013, red) and the 2 days thereafter (11 and 12 June 2013, orange and yellow). Note that the Harrier followed strips of mown habitat.

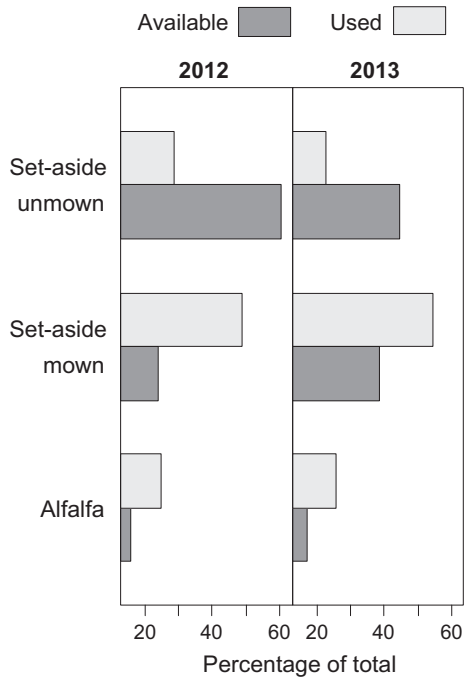


Figure 4. Habitat use of Montagu's Harriers hunting on 'Birdfields' compared with habitat availability.

DISCUSSION

Our results suggest that prey availability is more important for habitat selection than prey abundance alone, emphasizing the importance not only of enhancing prey densities but also of making this prey available. This result implies that it is important to measure prey availability (or hunting success) instead of prey abundance when evaluating the efficiency of AES.

The aim of this study was to show that the 'Birdfield' concept works in the sense that Harriers immediately respond when voles are made available, rather than quantifying the overall importance of 'Birdfields' for the reproduction or population growth of Harriers. The latter would require applying 'Birdfields' at larger spatial scales, comparing individuals breeding in areas with and without 'Birdfields'. This might be difficult as Montagu's Harriers in The Netherlands generally forego breeding in areas without AES. Nevertheless, evaluating the importance of 'Birdfields' for successful reproduction and population growth is an important next step, as we could imagine that 'Birdfields' are most effective in areas where

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Harriers strongly depend on small mammals (e.g. in France: Millon *et al.* 2008, Millon & Bretagnolle 2008) and less effective in regions in which Harriers feed mainly on birds (e.g. Terraube & Arroyo 2011).

'Birdfields' were not only favourable for Montagu's Harriers. During the breeding season, other vole-eating species such as Common Buzzard *Buteo buteo*, Common Kestrel *Falco tinnunculus*, Western Marsh Harrier *Circus aeruginosus* and Hen Harrier *Circus cyaneus* regularly visited 'Birdfields' for hunting. Furthermore, substantial numbers of Skylarks *Alauda arvensis*, a species that has heavily declined in our study area (Ottens *et al.* 2013), were nesting in the 'Birdfields'. This is important as Skylarks generally avoid nesting in AES, making it difficult to preserve this species in agricultural landscapes (Kuiper *et al.* 2013, 2015). Other passerines may profit from breeding in set-aside, and thus mowing part of the set-aside during the breeding season is not recommended. Finally, in winter, 'Birdfields' were a magnet for passerines such as finches and buntings, which fed on the cereal-rich set-aside strips, and wintering raptors such as Rough-legged Buzzard *Buteo lagopus* and Hen Harrier.

An important additional advantage of combining set-aside with a harvestable crop is that this makes the measure interesting from an economic point of view, for both the conservationist and the farmer. The current cost to realize 1 ha of set-aside is €2100, which is roughly equal to what a farmer would gain from growing winter wheat. As 'Birdfields' consist of a mixture of set-aside and alfalfa, less is eventually paid per hectare of 'Birdfield'. For example, the current configuration of 'Birdfields' consists of 40% set-aside and 60% alfalfa, so the cost is only €840/ha (2100*0.4). Thus, 'Birdfields' are a cheaper AES than set-aside without alfalfa strips; more hectares of 'Birdfields' can be realized with the same amount of money (in the current example, 2.5 times as much). Alfalfa was harvested and collected by the local crop drier (B.V. Oldambt, Oostwold, The Netherlands) and used for the production of commercial animal food pellets. A farmer normally earns €750/ha from growing alfalfa. It is thus highly advantageous for a farmer to combine alfalfa with set-aside, as the profit from growing set-aside (€2100/ha) more than outweighs the loss of the production of alfalfa (€750/ha), and there might be an additional small financial gain if part of the set-aside is also harvested (in this particular case, half of the set-aside was harvested during the second alfalfa harvest). Altogether, the farmer earns €1290/ha. It should be stressed that 'Birdfields' are not a profitable alternative to winter wheat (from the perspective of the farmer) as the profit of growing wheat is €2100/ha.

In a world that needs to feed an ever-expanding human population, there will be increasing pressure on biodiversity, and hence the cost efficiency and ecological effectiveness of AES need to be improved. We advocate that AES can be more efficient when measures are designed according to the behaviour and specific ecological requirements of a target species. We have shown that combining areas with high food availability and making these available through common agricultural practices is a promising approach. This is only possible when detailed ecological knowledge of the target species is available, and implementation of AES should therefore always be combined with monitoring and research activities, aiming at a more adaptive conservation approach.

We thank the Alfalfa harvesting company *Groenvoederdrogerij Oldambt B.V.* for their collaboration and all farmers in our study area for giving permission to count vole burrows on their fields, especially Z. Krabben-Renken. For help with fieldwork we would like to thank H. Bouman, M. Dijksterhuis, R. Erens, J. Gros, E. Karhausen, P. Kruiger, M. Kuiper, M. v. Laar, M. Meulman, H.J. Ottens, M. Postma, M. Pot, D. Schoppers, S. Van der Sijs, S. Sipma, O. Vlaanderen and P. Wiersma. M. Pot helped with entering the field data. This study was financed by the Ministry of Economic Affairs (EZ), the province of Groningen and Prins Bernhard Cultuurfonds. Our tracking studies are facilitated by the UvA-BiTS virtual lab (www.UvA-BiTS.nl/virtual-lab), an infrastructure for e-Science developed with support of the NLeSC (http://www.esciencecenter.com/) and LifeWatch, carried out on the Dutch national e-infrastructure with the support of the SURF Foundation. We thank R. Kimball, J. Martínez-Padilla and two anonymous referees for valuable comments on an earlier draft of the manuscript.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Figure S1. Location of the 'Birdfields' within the study area.

Figure S2. Distribution of mown alfalfa, mown set-aside and unmown set-aside after the first mowing events in 2012 and 2013.

Figure S3. Map of the locations of the vole burrow transect counts throughout East Groningen. Yellow boxes indicate the location of the 'Birdfields'.

Figure S4. Use of 'Birdfields' by Montagu's Harriers in relation to the distance from their nest.

Table S1. Seed mixture for set-aside strips.

Table S2. Vole abundance (mean number of vole burrows per field counted on six transects of 100 m length) in different crops in 2012 and 2013. Given are median, mean, minimum and maximum values and number of fields sampled (*n*).

Table S3. *P*-values of multiple comparison chi-square tests of the number of vole burrows in different crops.

Table S4. Number of vole activity signs (vole burrows, runways and food collection places) in 1-m^2 plots on 'Birdfields' in mown alfalfa strips (left) and mown set-aside strips (right) in 2012 and 2013. Given are mean, minimum and maximum values and number of plots counted (*n*) separated for the two fields and two mowing events.

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Minsmere Flood Risk Management Scheme

working with nature to adapt to climate change was the most successful and sustainable approach in this environmentally-sensitive coastal area

by Alex Schofield & Ed Ferguson

Minsmere Nature Reserve is owned and run by the Royal Society for the Protection of Birds (RSPB) in Suffolk, England. The site is internationally designated as a Special Protection Area, Special Area of Conservation, and Ramsar site (internationally important wetland); as well as being nationally designated as a Site of Special Scientific Interest and Area of Outstanding Natural Beauty. The coastal frontage (seaward of the coastal defences) is important for its coastal features, whilst the area landward of the coastal defences is important for its freshwater habitats and species.



View of the penstock controls, where timber fencing provided safety and aesthetically blended into the natural landscape of the Minsmere Royal Society for the Protection of Birds (RSPB) reserve - Courtesy of Black & Veatch

Understanding the need

To maintain the welfare of the public and our natural areas we are often required to try and exert control over natural forces. But resisting coastal change through hard-engineering is often not the most sustainable solution. Instead, there is a growing awareness of the need for engineering solutions to work with nature.

The project team and stakeholders understood this need in the planning, design and implementation of a scheme to help protect an internationally important area of nature conservation lying on one of the UK's fastest-eroding coastlines. The Minsmere Flood Risk Management Scheme allows natural coastal change to occur while protecting the majority of the freshwater features of the nature reserve from the sea.

Existing defences

The coastal defences consist of a frontline dune system (primary bank). A tidal sluice and outfall approximately midway along the frontage drains the hinterland via gravity. A man made secondary bank is located behind the primary bank north of the sluice. The sea defences have been under increasing pressure from the sea, with the primary sea defence being breached during a surge in 2006. Simply maintaining the pre-existing sea defence system would mean that coastal habitats would be increasingly subjected to coastal squeeze, as sea levels rise in the future. Allowing the coastal defence to fail would lead to the loss of the internationally important freshwater site.

The Environment Agency had previously engaged Black & Veatch to investigate flood management options for the Minsmere coastal frontage when the existing flood defences ceased to provide an adequate standard of protection due to coastal erosion and sea level rise. Black & Veatch had also developed the business case for the selected solution; which was largely driven by the need to defend the internationally protected habitats and species at the site from deterioration (a requirement of the Habitats Directive).

Preferred solution

In April 2010, the Environment Agency commissioned the preferred solution which was identified through close working with Natural England, the RSPB and other stakeholders. The solution was to allow the most at-risk coastal defences to continue to evolve under natural processes in one area – for which 28 hectares of replacement 'Compensatory' habitat has been included as part of the project –

whilst protecting the majority of the freshwater site against coastal flooding. An existing earth embankment, known as the Coney Hill Cross Bank, separates two areas of the Minsmere site.

This was rebuilt to provide a controlled cut-off between the two areas and therefore an improved line of defence from coastal flooding within the RSPB reserve. A new culvert was constructed through the bank and an associated water control structure was installed. Two coastal bunds were also required to help reduce erosion on the coastal frontage.

Undertakings

The Environment Agency led and funded by the £1m sea defences project with. Black & Veatch providing detailed design and site supervision for the scheme. This included environmental surveys; preparing and submitting the planning application (which included an Environmental Report to document the Environmental Impact Assessment); and preparing the contract document pack for tender and construction.

For the construction phase of the works, Black & Veatch continued to provide design and environmental support to the team, undertaking the roles of Site Supervisor and Environmental Clerk of Works. Interserve was the scheme's principal contractor.

Rising to the challenges

Project challenges were threefold: a large volume of construction traffic to a remote location; the technical problems of soft and very wet ground conditions; and the number of protected species in the area.

Construction traffic: The remote location and popularity of Minsmere posed a substantial design and construction challenge. More than 90,000 visitors to the reserve annually use a single-track road, which passes through villages and becomes a footpath for the final 3km. After concluding that on-site fill material for the flood-defence embankment was not an option and despite reducing the required volume by a third through reuse where possible, the project team realized that importing fill material to the remote rural site would disturb the local community as well as the RSPB reserve and would pose safety risk associated with public interaction with site vehicles and plant. The team focused on risk mitigation through effective coordination, planning and communication.

Early engagement with stakeholders enabled the project team to identify the best site route and constraints. With the RSPB, the team implemented an alternative route for a footpath along – but safely removed from – the reserve's access road. Traffic management discussion with the local community through the Parish Council and public forum identified additional constraints, such as important local events. The team developed design-stage traffic management plan (TMP), which was disseminated as a national best practice for projects with similar issues. The contractors further developed this in construction-phase TMPs with their tender returns. Close work with the RSPB and distribution of community newsletters helped foster understanding and acceptance.

Success can be measured not only by the complete absence of accidents and complaints but also by multiple positive comments and an RSPB reserve visitor's tweet that said it all: "*If you want to see best-practice traffic management, come to Minsmere*."

Ground conditions: The new flood embankment was constructed within a reed bed. The soft and wet ground presented a technical and construction challenge that was addressed through various measures. Alignment of the new flood bank was designed to take advantage of better areas of ground, and the foundation and toe detail of the bank were designed to avoid excavation into the soft ground; geotextile and hard core layers were used to create a firm foundation. A partnered approach led to a construction



Minsmere coastal area, designated internationally as a Special Protection Area, a Special Area of Conservation and Ramsar Wetland
Courtesy of Black & Veatch

methodology that consisted of creating a haul road from an existing raised footpath that later formed part of the new embankment. This provided an access route across the reed bed and also precluded the need for working platforms within the reed bed.

Species protection: While the goal of the project was to protect as much freshwater habitat as possible, everyone involved with the project was keenly aware of the need to protect the reserve's many rare and protected species of plants and animals during construction. To this end project personnel worked closely with RSPB to carry out species surveys, design ecological mitigation measures, reduce and manage water levels, and utilise the most appropriate plant and equipment (for example, an amphibious excavator was used to undertake works in the reed bed).

Restricting the timing of work to outside the bird breeding season required careful advanced planning and rapid construction activity. Working hours and materials delivery were scheduled so as to minimize disturbance to birds and bats as well as school collections and drop-offs. The works commenced in September 2011 and were completed by January 2012. Although winter is not the ideal time for construction, activities were planned to make the best use of favourable weather.

An environmental scientist on the project team developed mitigation plans for the main protected species of flora and fauna. The scientist also facilitated mitigation of an invasive plant species to avoid spreading. 'Species management cards' produced in conjunction with the contractor's environmental representative informed site workers what to look for and what to do upon spotting these species.

Other special measures to protect wildlife included the following:

- Staging of vegetation clearance prior to construction to encourage water voles to voluntarily vacate the working areas.
- Capture and relocation of reptiles to other parts of the Minsmere Nature Reserve prior to topsoil stripping.
- Fish relocation (through electro-fishing and capture-release), to a non-affected area of the site, prior to lowering of water levels.
- Design and construction of a bespoke eel pass for the new water control structure conforming to the RSPB's specific operational requirement.
- Well-considered site compound and access route fencing to minimise disruption to the range of species present (for example to allow continued deer crossings).

Additional project highlights and acclaim

The works were carefully designed with full consultation with the RSPB to ensure they blended well with the reserve. For example, sand was used to cover the clay coastal embankment, then existing vegetation was put back into place to provide a more natural appearance.

Site waste minimization, environmental action plans and a carbon calculator provided direction and benchmarks, and demonstrated cost savings of £142,000 as well as a carbon saving of 43%.

The project has earned multiple awards in recognition of environmental and project safety achievements.

- Gold-level Green Apple Award in November 2012. This national campaign recognizes Britain's greenest companies, councils and communities for Environmental Best Practice. The Green Apple Awards honour environmental endeavour, and the Gold award is the highest accolade bestowed by the Green Organisation.
- Earlier last year the Minsmere project was credited across all four categories; physical achievement, sustainability, team achievement, and technical excellence and innovation, with the Institution of Civil Engineers (ICE) East of England Merit Award.
- The Association for Project Safety (APS) awarded the scheme 'Engineering Project of the Year' at its Annual Awards 2012. The APS awards recognize excellence in health and safety risk management in the construction industry. The judges cited "*excellent early cooperation between designers and contractors*" and the "*huge effort… to protect the public during construction*" as award-worthy achievements.

Conclusion

The construction works were successfully completed in January 2012, slightly ahead of schedule and under budget. A truly collaborative effort has brought to conclusion a scheme of significant environment value. Despite a myriad challenges, project participants successfully built a new coastal defence that will continue to mitigate natural but destructive forces and protect the 246 hectare internationally treasured environmental and landscape asset, along with 12 homes, for the next 50 years.

Early planning allowed project participants to work within a special area heavily populated by wildlife and visited by admirers with minimum impact to people and the environment. Effective overall management, a partnered team with shared vision, and engagement with the local community were some of the factors that contributed to the success of the Minsmere flood risk management scheme.

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Narrow access road, depicting one of the most severe project challenges and requiring careful traffic management planning and effective public communication - Courtesy of Black & Veatch



Visiting children from the local primary school with their painted observations, as part of the community education and outreach programme - Courtesy of Black & Veatch

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Effects of ship traffic on seabirds in offshore waters: implications for marine conservation and spatial planning

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Abstract. Most anthropogenic influences on marine ecosystems, except for river- or terrestrial-borne pollution, involve some sort of vessel activity. Increasing anthropogenic activities mean that many countries are being forced to develop spatial planning schemes, while at the same time implementing conservation sites for sensitive species at sea. The effects of ship traffic on seabirds sensitive to human disturbance are currently too poorly understood to allow for the development of proper planning and conservation guidelines. We therefore used aerial surveys and experimental disturbance to elucidate the effects of passing ships on the distribution patterns, habitat loss, and species-specific flight reactions of birds, as well as the potential for habituation. Loons (*Gavia* spp.) showed clear avoidance of areas with high shipping intensity. Flush distances of four sea duck species differed significantly, with the longest distances recorded for Common Scoters (*Melanitta nigra*) and the shortest for Common Eiders (*Somateria mollissima*). Flush distance was positively related to flock size. Among all the sea duck species studied, the duration of temporary habitat loss was longest for Common Scoters. We found indications of habituation in sea ducks within areas of channeled traffic. However, it is questionable if habituation to free-ranging ships is likely to occur, because of their unpredictable nature. We therefore recommend that spatial planning should aim to channel ship traffic wherever possible to avoid further habitat fragmentation and to allow for habituation, at least in some species. Information on the effects of shipping on other seabird species and during different periods of the year is urgently needed, together with information on the effects of different types of boats, including recreational and fishing vessels.

Key words: *Black-throated Loon; Common Eider; Common Scoter; distribution; disturbance; German North Sea and Baltic Sea; habitat loss; Long-tailed Duck; Marine Special Protection Area; Red-throated Loon; shipping; White-winged Scoter.*

Introduction

Spatial planning of anthropogenic activities in the offshore zone is currently in progress in many countries worldwide. Most activities in marine ecosystems are in some way related to shipping. Disturbance by ships can be a major threat to birds, through affects on behavior, reproduction, and fitness of individuals in colonies (e.g., Burger 1998, see review in Carney and Sydeman 1999), as well as on foraging or resting habitats (e.g., Rodgers and Schwikert 2003, Stolen 2003, Kaiser et al. 2006). However, the effects of ship traffic on seabirds are still too poorly understood to allow for either proper spatial planning or conservation management actions.

The situation in the German offshore zone can be regarded as typical of that in many countries that are currently forced to combine anthropogenic activities (such as shipping) and conservation needs in their planning schemes. The German North Sea is one of the most intensely used marine systems worldwide (Halpern et al. 2008), and shipping is particularly intense in the German North Sea and Baltic Sea (Knust et al. 2003). These bodies of water are used as resting and foraging sites by high numbers of seabirds that are sensitive to human disturbance, many of which are listed in Annex I of the European Union (EU) Birds Directive (e.g., Garthe et al. 2003, 2007, Mendel et al. 2008). Special Protection Areas (SPAs) have recently been implemented in the Exclusive Economic Zones under the EU Birds Directive (Garthe 2006; hatched areas in Fig. 1b).

Numerous studies have examined the conflict between birds and ships in freshwater sites (e.g., Tuite et al. 1983, Rodgers and Schwikert 2003). In contrast, basic information on the impacts of ships on seabirds in offshore waters is generally scarce. The few studies involving marine habitats have focused on single species in connection with cargo shipping (Kube 1996, Camphuysen et al. 1999, Kaiser et al. 2006). Basic information on the effects of passing ships on individual flight reactions, distribution patterns, and habitat loss for a suite of sensitive species in a marine system has not yet been quantified. However, such basic knowledge is

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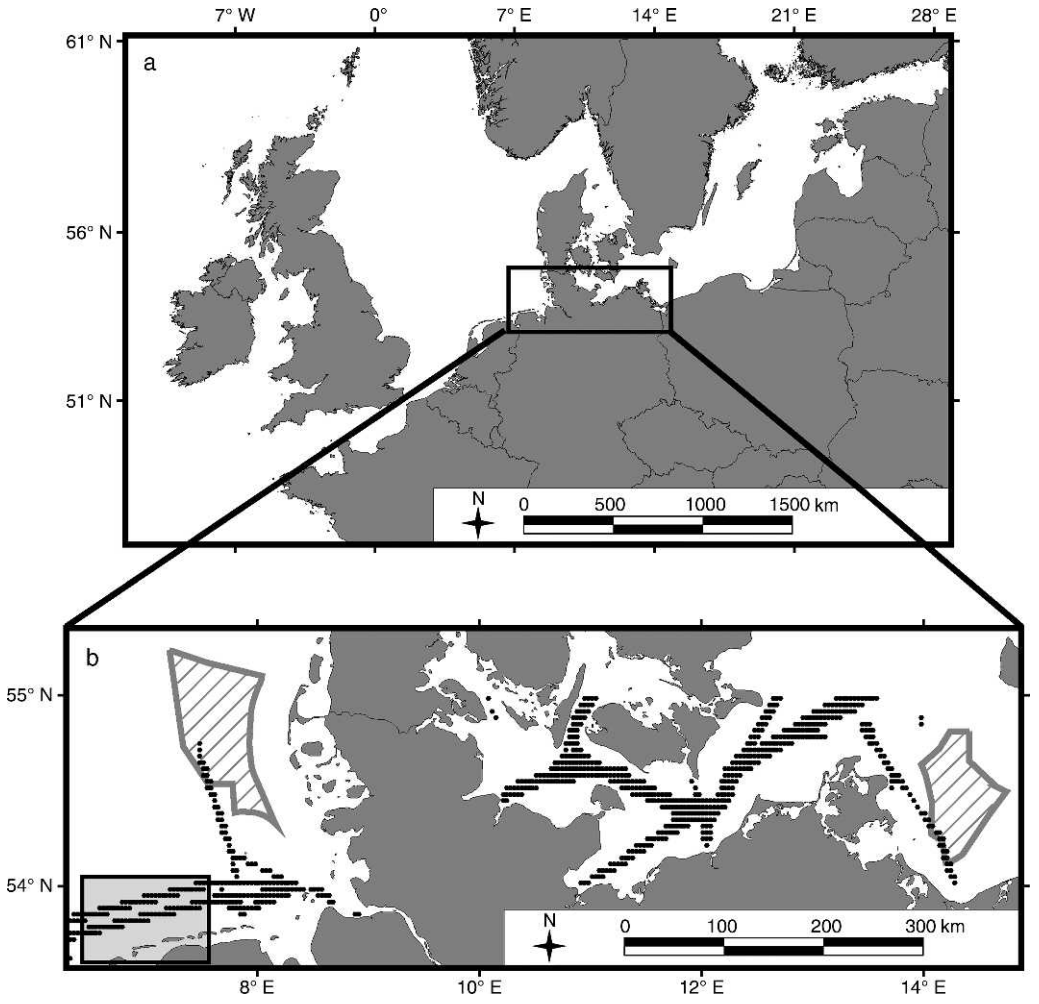


FIG. 1. Study area in the German parts of the North Sea and Baltic Sea, indicating the location of shipping lanes (dots indicate >10 merchant ships passing per day; due to overlap, in some cases, the dots resemble bold lines; Knust et al. 2003). The light-gray rectangle in panel (b) indicates the area used for aerial counts to analyze the effects of ship traffic on distribution of loons (*Gavia* spp.) across a commercial shipping lane system; hatched areas are Special Protection Areas (SPAs).

essential for appropriate marine spatial planning and conservation management.

The aims of this study were therefore to: (1) test if ship traffic affects the distribution patterns of seabirds; (2) evaluate species-specific flush distances of sensitive seabirds when approached by a ship, and unravel the effects of a suite of environmental parameters on flight behavior; (3) estimate how long a disturbed patch is avoided for, i.e., temporal habitat loss; and (4) examine the possibilities for habituation of seabirds in relation to spatially restricted ship traffic.

For this study, we chose seabird species that have already been described as highly sensitive to shipping activities (Garthe and Hüppop 2004, Bellebaum et al. 2006, Kaiser et al. 2006, Mendel et al. 2008), i.e., Red-throated and Black-throated Loon (*Gavia stellata*; *G. arctica*), Common Eider (*Somateria mollissima*), Long-tailed Duck (*Clangula hyemalis*), Common Scoter (*Melanitta nigra*) and White-winged Scoter (*M. fusca*).

METHODS

Study area

The study was conducted in the offshore zone of the German North Sea (south of 54.2° N; east of 7° E) and Baltic Sea (south of 55° N; west of 15° E; Fig. 1). Two SPAs were established in the North Sea and Baltic Sea in 2004, respectively, under the EU Birds Directive (BMU 2004, Garthe 2006; hatched areas in Fig. 1b). The effects of ship traffic on the distribution patterns of Red-throated and Black-throated Loons were studied across a highly frequented shipping lane in the southern North Sea (light-gray rectangle in Fig. 1b). Species-specific flush distances, habituation, and temporal habitat loss were assessed for all four sea duck species in the Baltic Sea.

Distribution of loons across a commercial shipping lane

Commercial shipping in the German North Sea is particularly intense and highly concentrated within the

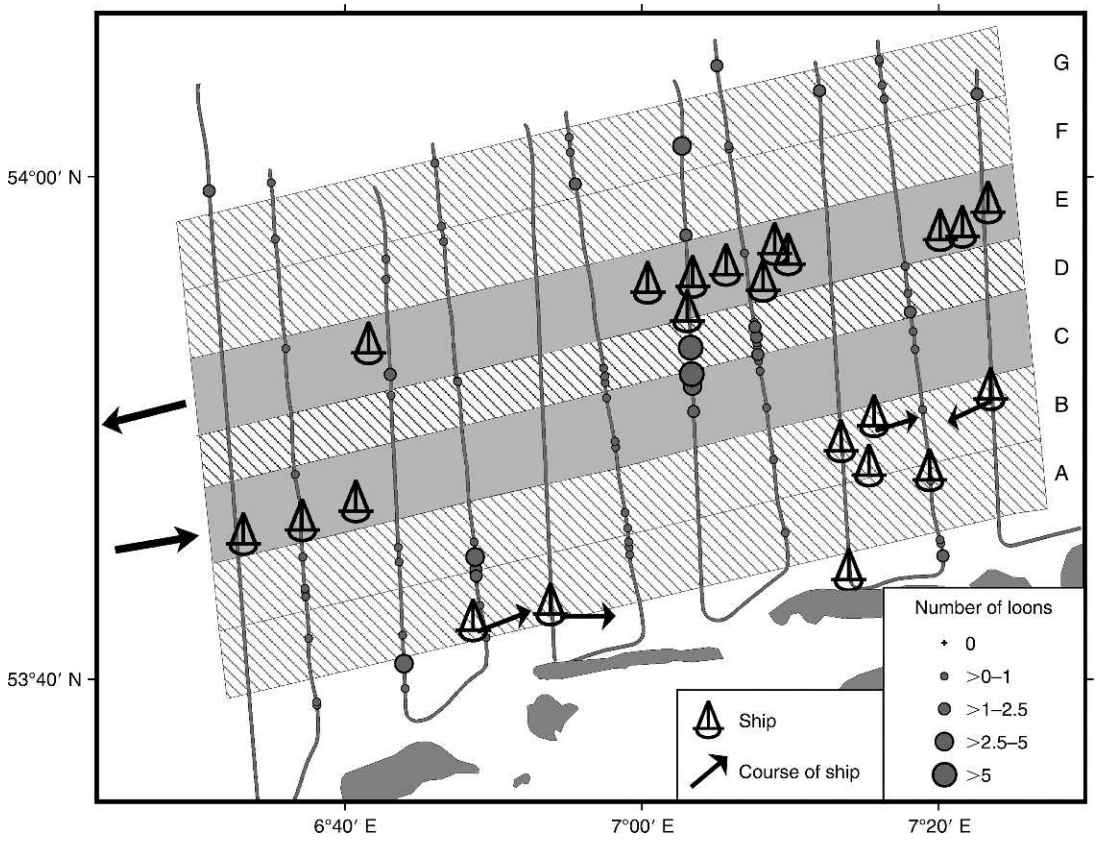


FIG. 2. Example of the distribution of loons recorded during an aerial survey flight in April 2006 (see light-gray rectangle in Fig. 1b) within seven zones (A–G) used for computing avoidance or preference values (see Table 1). Light gray zones (C and E) are areas of intense channeled shipping; diagonally hatched zones are areas of lower shipping intensity. Positions and (in several cases) courses of the ships recorded during the survey are shown.

shipping lanes in the southern area (Knust et al. 2003; Fig. 2). This area provides the opportunity to study the effects of channeled ship traffic on seabirds. All commercial ships must sail within a northern shipping lane when travelling west and in a southern lane when travelling east (zone C = 5.7 km and zone E = 5.5 km wide, respectively; see arrows in Fig. 2). These two shipping lanes are separated by zone D (3.8 km wide). Shipping intensity is particularly low in this zone, as vessels may only cross at right angles, and crossing rarely occurs. Both the shipping lanes and the separation zone exhibit the same water depth as the surrounding waters; however, fishing is prohibited in these areas, whereas it is allowed in the surrounding waters. Smaller, noncommercial ships and recreational vessels are allowed to sail inshore of the eastbound shipping lane in any direction (bands A and B in Fig. 2, each 5 km wide). Shipping is unrestricted north of the westbound shipping lane (bands F and G in Fig. 2, both 5 km wide). The overall numbers of ships are generally much higher within the shipping lanes (C and E) than in all other zones.

Because of their sensitive nature (Garthe and Hüppop 2004, Mendel et al. 2008), we hypothesized that loons would avoid the two shipping lanes as a consequence of disturbance by intense, channeled shipping. Data for Red-throated and Black-throated Loons were pooled, because it was not possible to reliably identify them to species level in aerial surveys. Both species are similarly sensitive to anthropogenic disturbances (e.g., Garthe and Hüppop 2004, Mendel et al. 2008). Avoidance of and preference for each of the seven zones were tested using the selectivity index of Jacobs (1974). In total, three survey flights (for a detailed description of the recording methods, see Diederichs et al. 2002) were conducted across the seven zones during March and April in 2002 and 2006 (for sample sizes, see Table 1). The distribution of loons was tested for homogeneity beforehand. No reduction in loon densities was detected with distance from coast; linear model (LM), $t = -0.003$, $df = 19$, $P = 0.77$.

Flush distances

Flush distances (defined as the linear distance from the observer vessel to the birds at the moment of take-off from the water) were measured for sea ducks on 15 survey days between October 2004 and April 2005. Loons were not included in these analyses because the

TABLE 1. Selectivity indices for loon (*Gavia* spp.) distribution during three survey flights across seven zones in the German North Sea located within the shipping separation zone.

Date	Area surveyed (km)	Loons (*n*)	Zone						
			A	B	C	D	E	F	G
March 2002†	437	19	−0.19	0.01	−1	0.46	−1	0.61	−0.11
March 2006	724	38	0.44	0.1	−0.56	0.54	−0.21	−1	−0.47
April 2006	564	155	−0.12	0.06	0.07	0.54	−0.7	−0.51	0.12

Notes: Two zones (C and E) were located within the shipping lanes, while the others were located outside the shipping lanes (see Fig. 2). Negative values indicate avoidance; positive values indicate preference for the respective zone (−1 = total avoidance; 1 = total preference).

† Coverage of the whole area was achieved during two consecutive survey days; data were recorded by Bioconsult SH, Husum, Germany.

sample sizes were too low. Three different research vessels were used: (1) *MS Haithabu* (25 m long; $n = 219$ measurements), (2) *MS Victor Hensen* (40 m long; $n = 138$ measurements), and (3) *MS Ludwig Prandtl* (31 m long; $n = 189$ measurements). All the ships were similar heights (5–6 m excluding masts). We approached the birds at a constant speed of 18 km/h (maximum possible speed of all vessels used). Only the flush distances of flocks of birds located within 300 m of either side of the ship's track line were measured, as we assumed that flocks too far outside the track line of the vessel would not show comparable flush reactions.

A total of 546 flush distances of single individuals or flocks of different sizes were measured (see Fig. 3a for sample sizes of the different species). Among these, 144 values represented birds that did not fly off or were estimated by eye, as the individuals only escaped when the approaching vessel was very close. A total of 202 flush distances were measured using binoculars (7 × 50, Fujinon, Willich, Germany) containing reticles, which allowed the angle between the birds on the water surface and the horizon to be measured (Lerczak and Hobbs 1998). In combination with observer altitude, this provided a measure of distance (Laake et al. 1994). The method of Heinemann (1981) was used for 200 measurements. This method is based on the same geometric principle as the above method, but using calipers instead of binoculars with reticles: Holding the calipers with the observer's arm outstretched, the upper arm of the calipers is positioned on the horizon and the lower arm on the flock of birds. With known observer altitude and arm length, the caliper reading provides the distance of the birds from the observer, using geometric functions. We did not use radar (compare Bellebaum et al. 2006, Kaiser et al. 2006) to measure distances, as this requires very calm conditions and can only be used effectively for flocks of many birds; because we were examining the influences of sea state and flock size on flush distance, it was necessary to estimate flush distances of single individuals and of small flocks during rough conditions. For both methods used, the water surface was constantly scanned using binoculars to ensure that smaller flocks at larger distances were included. We tested for differences between flush distances acquired by the two different methods for each sea duck species. The two methods were not used

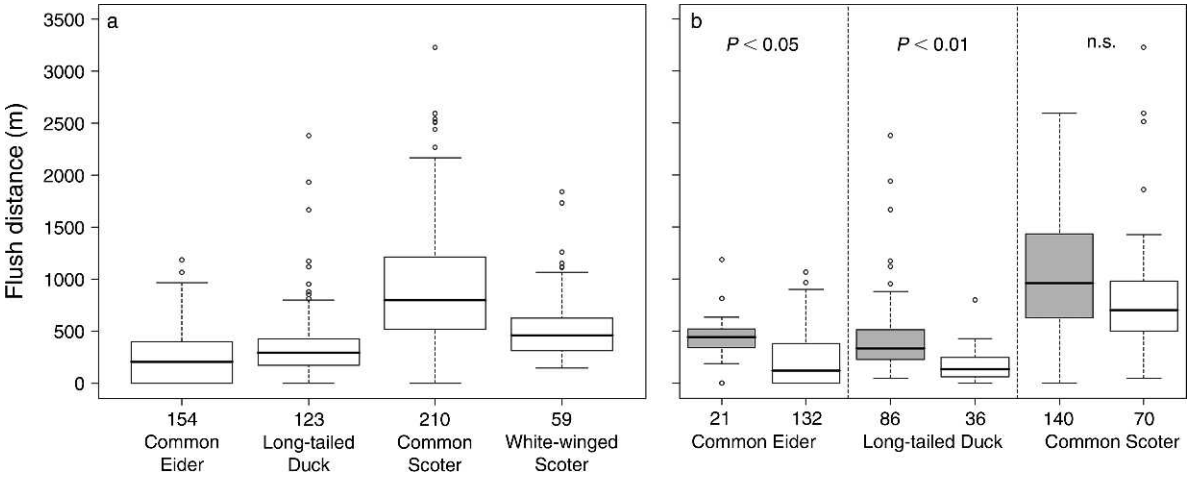


FIG. 3. Flush distances of sea ducks in front of the approaching vessel: (a) for all measurements and (b) separated for measurements outside shipping lanes (gray boxes) and within shipping lanes (open boxes). The bold lines represent the median; boxes are 25% confidence intervals; whiskers are 95% confidence intervals; and dots are outliers. Numbers below the columns depict the sample sizes; "n.s." indicates not significant. Statistical tests were corrected after Bonferroni.

simultaneously for the same flocks of birds. However, all parameters remained similar during all cruises: ship speed (18 km/h in most cases; range = 16.7–20.3 km/h); observer (always the same individual); weather conditions (no rain; wind speed range = 12–38 km/h). The sample sizes for both methods were large and the measurement methods were applied alternately. We therefore considered the comparison between the methods to be reliable, even though they were not tested on exactly the same flocks. The LM detected no significant differences in flush distances between the two methods for any species (Common Eider: $t = 0.4$, df = 72, $P = 0.73$; Long-tailed Duck: $t = -1.14$, df = 100, $P = 0.26$; Common Scoter: $t = 1.96$, df = 207; $P = 0.06$; White-winged Scoter: $t = -0.16$, df = 56; $P = 0.87$). Data acquired using both methods were therefore pooled.

Data on flush distances were left-skewed, and they were, therefore, log-transformed, which resulted in a sufficiently normal distribution of the residuals. All analyses of flush distances were therefore performed using LM analysis. This approach was preferred to the use of a general LM with non-normal errors, because it allowed easier interpretation of the modeled parameters. Differences in flush distances among the four duck species were examined using LM (e.g., Chambers 1992) with the Bonferroni correction. The influences of six environmental variables (predictor variables) on flush distances were also modeled using LM analysis: (1) water depth, (2) flock size, (3) state of the sea surface according to an eight-fold relative score following Garthe et al. (2002), (4) type of observer vessel (differences in shape or engine noise may affect flight behavior), (5) month of observation (differences in molting stage may affect take-off capabilities), and (6) location of the flock within or outside a commercial shipping lane. The last parameter was included in the model to take account of possible habituation to temporally and spatially regular ship traffic. Data were classified on the basis of individuals that were located within the track or within 2 km distance of commercial shipping lanes (all such situations are subsequently referred to as "within shipping lanes"), and individuals farther away from shipping lanes (subsequently referred to as "outside shipping lanes"; source Knust et al. 2003). The six predictor variables were standardized to account for differences in their scales. Both linear and quadratic terms were considered for the parameters water depth, flock size, and sea state. Selection of the best model was based on backward selection: The first model used all predictor variables. The predictors that were not significant were then removed from the model step-by-step, starting with the variable with the highest P value. Model improvement was checked using AIC, as well as by comparing both models using analysis of variance (Venables and Ripley 2002). Predictors were removed sequentially until a minimum adequate model was produced. All LMs were performed using the open-source software package R 2.8.1 (R Development Core Team 2007). A linear mixed-effects model was used to test if the data were spatially autocorrelated, using the corExp function in the package nlme (Pinheiro et al. 2008). This revealed no spatial autocorrelation of flush distance data.

Temporary habitat loss

To estimate the time for which a disturbed habitat patch was lost for feeding or resting, numbers of the four sea duck species were recorded using the method for counting seabirds at sea described by Garthe et al. (2002) and Tasker et al. (1984). The observer vessel sailed along a predefined route while seabird counts were carried out. After a defined period of time (90 min or 60 min), the vessel turned and sailed back on the same course. The abundance values for each species before the first disturbance by the observer vessel were compared with those a certain time after the disturbance (numbers of individuals after the disturbance were expressed as a proportion of numbers of individuals before the disturbance).

Results

Distribution of loons across a commercial shipping lane

The three survey flights across the seven zones covering the shipping lane system in the North Sea clearly showed that loons avoided strips C and E (except for strip E during April 2006), where merchant shipping is channeled (Table 1). In contrast, the zone separating the shipping lanes (zone D) was preferred (Table 1; Fig. 2 provides an example of the distribution of 155 loons and 22 ships recorded in April 2006). Index values for the southernmost and northernmost bands were inconsistent among the three survey flights (Table 1).

Flush distances

All sea ducks that escaped in front of the approaching vessel took flight, except for a single Common Eider that dived to escape. Flush distances of Common Eiders (CE), Long-tailed Ducks (LTD), Common Scoters (CS) and White-winged Scoters (WWS) differed significantly from each other, according to LM using the Bonferroni correction: CE vs. LTD, $t = 5.4$, df = 275; CE vs. CS, $t = 14.3$, df = 362; CE vs. WWS, $t = 6.3$, df = 211; LTD vs. CS, $t = 10.8$, df = 331; LTD vs. VS, $t = 3.9$, df = 180; CS vs. WWS, $t = -5.1$, df = 267; $P < 0.001$ for all cases; Fig. 3a).

Flush distances were highest for common scoters (median, 804 m), followed by White-winged Scoters (404 m), Long-tailed Ducks (293 m), and Common Eiders (208 m; Fig. 3a). Flush distances were highly variable within species (particularly in Common Scoters), indicating strong inter-individual variability in reactions. The most extreme example involved one Common Scoter flock that flushed at a distance of 3.2 km. In contrast, several flocks (mainly of Common Eiders) did not fly in front of the approaching vessel, but remained on the sea surface: 29% of all flocks of Common Eiders,

TABLE 2. Output of the linear model relating flush distance to environmental parameters.

Parameter	Common Eider		Long-tailed Duck		Common Scoter		White-winged Scoter	
	t	*P*	*t*	*P*	*t*	*P*	*t*	*P*
Flock size	3.12	**0.002**	4.18	**<0.001**	9.89	**<0.001**	n.s.	n.s.
Flock size (sq.)	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	−2.70	**0.009**
Shipping lane	−2.65	**0.009**	−3.10	**0.003**	n.s.	n.s.	n.t.	n.t.
Water depth (m)	n.s.	n.s.	n.s.	n.s.	−2.49	**0.013**	n.t.	n.t.
Water depth (m) (sq.)	n.s.	n.s.	n.s.	n.s.	2.53	**0.012**	n.t.	n.t.
Sea state	2.38	**0.019**	2.90	**0.005**	−3.46	**<0.001**	n.s.	n.s.
Sea state (sq.)	2.49	**0.014**	−2.63	**0.010**	n.s.	n.s.	−2.60	**0.012**
Observer vessel	n.t.	n.t.	n.s.	n.s.	n.s.	n.s.	1.89	*0.064*
Month	n.s.	n.s.	5.51	**<0.001**	n.s.	n.s.	n.t.	n.t.

Notes: For several parameters, both linear and square effects (sq.) were taken into account. Several parameters could not be tested (n.t.) because of inadequate sample sizes. Significant effects ($P < 0.05$) are indicated in boldface; italics indicate $P < 0.1$; and "n.s." indicates not significant. Sample size, degrees of freedom, and R^2 for each species are as follows: Common Eider, $n = 154$, df $= 148$, $R^2 = 0.15$; Long-tailed Duck, $n = 123$, df $= 116$, $R^2 = 0.44$; Common Scoter, $n = 210$, df $= 205$, $R^2 = 0.33$; White-winged Scoter, $n = 59$, df $= 55$, $R^2 = 0.01$. R^2 values represent multiple correlation between flush distance and all of the environmental parameters.

5% of Long-tailed Ducks, 0.5 % of Common Scoters, and 0 % of White-winged Scoters did not take flight.

Environmental parameters were able to partially explain the variability in flush distances. Flock size had the strongest effect, at least for Common Scoters and Long-tailed Ducks (Table 2). Flush distances were positively correlated with flock size (Fig. 4) in all species, except White-winged Scoter, which showed no correlation. Flock size was highest for Common Scoters (median = 8 birds; maximum = 500 birds) and in Long-tailed Ducks (median = 5.5 birds; maximum = 180 birds), followed by Common Eiders (median = 5 birds;

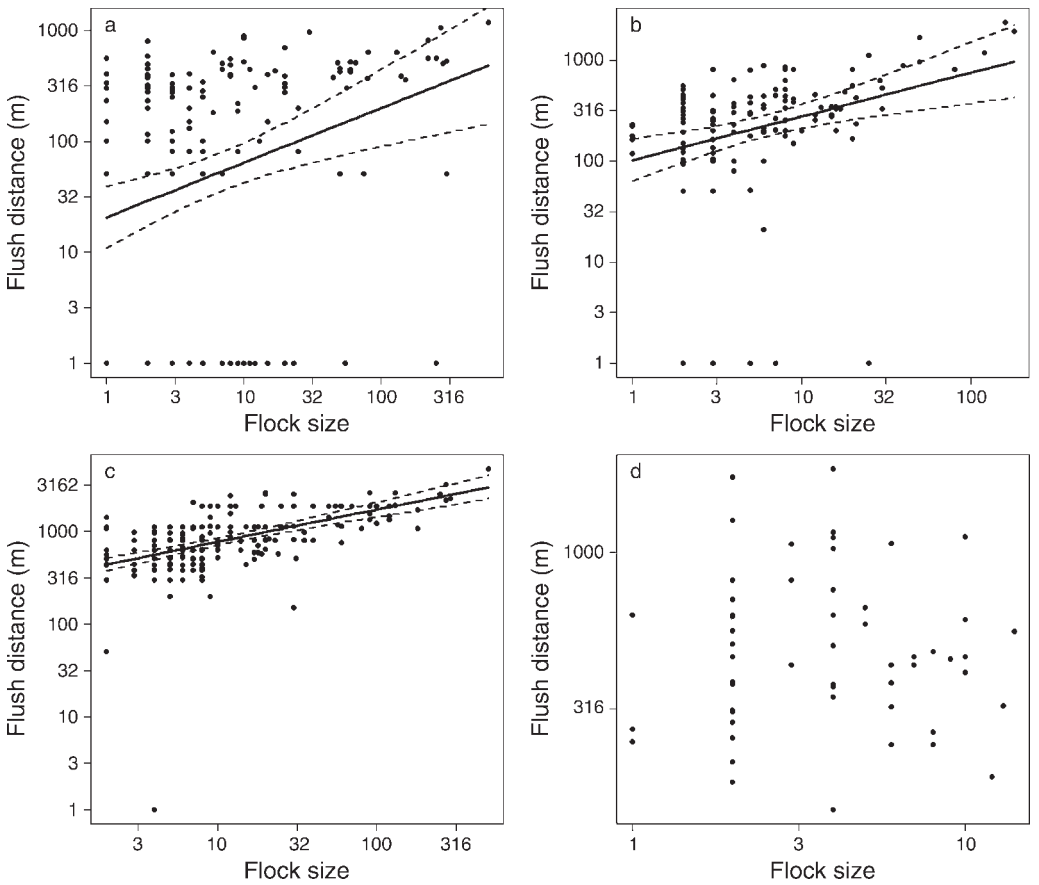


FIG. 4. Flush distances of (a) Common Eiders, (b) Long-tailed Ducks, (c) Common Scoters, and (d) White-winged Scoters in relation to flock size. The solid line is the linear model of flush distance; dashed lines indicate the 95% confidence interval of the model. Note differences in scale of the axes between the species.

maximum = 720 birds), while White-winged Scoters showed the smallest flock sizes (median = 4 birds; maximum = 14 birds). Flocks of >100 individuals were generally scarce for all species. Flock location (i.e., within or outside shipping lanes) had a significant effect on flush distance; flush distances were lower within than outside shipping lanes for both Common Eiders and Long-tailed Ducks (Table 2, Fig. 3b). The importance of the parameter "location of the flock" was emphasized by the fact that 96% of the Common Eider and 100% of the Long-tailed Duck flocks that did not take flight were located within shipping lanes. Sea state also had a strong influence on flush distance; most species flushed at shorter distances with increasing sea state. In contrast, the type of observer vessel and water depth had little or no effect on flush distances. The observation month only had a strong effect on flush distance in Long-tailed Ducks (Table 2).

Temporary habitat loss

For all species, individual numbers were lowest immediately after the disturbance (Table 3). Common Eider and Long-tailed Duck numbers were relatively higher during the first hour after the disturbance, and increased relatively rapidly in subsequent hours, compared with the other species. The increase in numbers was much slower for White-winged Scooter, while Common Scoters did not reach pre-disturbance numbers until after the experiment had finished (Table 3).

Table 3. Temporary habitat loss by sea ducks.

Species	Disturbed individuals (%)		
	1–60 min	61–120 min	121–180 min
Common Eider	17	107	...
Long-tailed Duck	26	57	113
Common Scoter	4	10	13
White-winged Scoter	3	24	66

Notes: Values indicate the percentage of individuals recorded 1–180 min after disturbance by the observer vessel relative to the number of individuals recorded before the disturbance. The ellipses indicate that the sample size was too low.

Discussion

Effects of ship traffic on loon distribution

The results of this study clearly demonstrate that ship traffic affected species-specific flight reactions in sea ducks, and also influenced the distribution of loons. Loon numbers in shipping lanes were significantly lower than in other areas (particularly compared with the zone of lowest shipping intensity). Food availability is a crucial factor in determining the presence or absence of a species in a certain sea area (e.g., Kube 1996, Kaiser et al. 2006, O'Connell et al. 2007). Thus, profitable feeding sites might be preferred despite high ship abundance, if the energetic costs of disturbance can be compensated for by more efficient feeding. However, at least in the case of piscivorous seabirds such as loons, the possibility that fish prey may also be disturbed by ships must also be considered (e.g., Misund and Aglen 1992), thus reducing effective foraging.

In contrast to sea ducks, loons are not dependent on shallow feeding grounds, because they are not restricted to shellfish banks and (at least in German waters) they are not bound to nearshore waters (Garthe 2006). Their distribution may therefore mirror the sites where conflict with shipping is most intense, as indicated by their avoidance of shipping lanes, compared to their preference for the shipping separation zone. In contrast to Common Eiders that showed shorter flush distances within shipping lanes, the avoidance of shipping lanes by loons suggests that habituation to passing ships is unlikely to occur in this group of species. This is supported by evidence that loons (in contrast to sea ducks) did not habituate to offshore wind turbines in Danish waters, even more than five years after construction (Petersen and Fox 2007, Petersen et al. 2008).

Flush distances

Very little information on flush distances of seabirds at sea is currently available in the literature. Using radar, Kaiser et al. (2006) estimated flush distances of common scoters ranging from 1000–2000 m, which were higher than those observed in the current study. There are two possible explanations for this discrepancy: (1) birds within different study sites may respond differently; and (2) the radar measurements performed by Kaiser et al. (2006) involved larger flocks, which we have shown exhibit longer flush distances. Bellebaum et al. (2006) measured flush distances of Long-tailed Ducks and White-winged Scoters in the Baltic Sea, using radar. Their findings for White-winged Scoter were comparable to our findings, while Long-tailed Ducks showed shorter flush distances than those in the current study. This difference might be explained by different environmental conditions, as Bellebaum et al. (2006) performed measurements on two single survey days, with sea states ranging from 0–1.

We found significant interspecific differences in flush distances among the four sea duck species investigated. These differences might be partially explained by differences in wing morphology and take-off capabilities among the species investigated: body mass and wing-loading is higher in Common Eiders than in the other sea duck species (Guillemette 1994, Guillemette and Ouellet 2005), while Common Scoters and Long-tailed Ducks have the lowest wing loadings. To save energy, heavy species with high wing loading should be more reluctant to take off, which could explain the comparatively short flush distances of Common Eiders (as well as the higher proportion of birds that did not take flight at all.

A high degree of the intraspecific variability in flush distances detected in this study could be well explained by differences in flock size and environmental variables:

TABLE 4. Summary of the main results for the species investigated.

Variable	Common Scoter	Long-tailed Duck	Common Eider
Flush distance	very high	moderate to high	low to moderate
Flock size	very high	moderate	moderate to high
Habituation to channeled traffic	no clear habituation found	indications for habituation found	indications for habituation found
Temporary habitat loss	high	low to moderate	low
Others	long period of extensive wing molt accompanied by period of flightlessness;§ molting aggregation during summer in eastern Baltic Sea¶ when recreational traffic is intense†		high proportion of birds located in shipping lanes; high proportion of birds did not take flight; absence of flight reactions may be linked to high wing loading#

† P. Schwemmer and S. Garthe, *personal observations*.
‡ Bellebaum et al. (2006).
§ Mendel et al. (2008).
¶ Sonntag et al. (2004).
# Guillemette (1994).

Flock size especially had a significant positive effect on the flush distances of all sea ducks, except for White-winged Scoters (which showed constantly low flock sizes). This could be explained by the fact that the whole flock took flight whenever the most sensitive individuals within the flock started to escape. The probability of encountering more sensitive individuals is higher in larger flocks, and thus, larger flocks will tend to flush at longer distances than smaller flocks or single individuals. We expected month of observation to affect flock size, as flocking behavior may differ between migration and non-migration periods. This could have had an indirect impact on flush distances. However, no such correlation was identified for any of the species investigated.

In addition to flock size, the location of the flock with respect to the shipping lanes was also important for several species. Individuals within shipping lanes showed shorter flush distances, suggesting that these individuals had become habituated to ship traffic. Other authors have detected habituation in birds that were frequently disturbed (e.g., Burger and Gochfeld 1991, Whittaker and Knight 1998). However, it is difficult to determine if weak responses by animals to disturbance should be classified as habituation. An important prerequisite for true habituation would be that individuals should show a certain degree of philopatry (i.e., staying in a certain area for a long enough time, e.g., because of a rich food supply). This information is currently lacking for the species studied, and the use of telemetry devices could represent the most effective way of studying this further (e.g., Kotzerka et al. 2010). However, long-term seabird surveys have shown that their large-scale distribution patterns remain stable over the seasons (e.g., Garthe et al. 2003), suggesting a certain degree of philopatry. Thus, the weaker response of individuals located in shipping lanes may indeed reflect habituation. The high proportion of common eider flocks located within shipping lanes and their significantly shorter flush distances compared with the other species suggests that habituation was strongest in this species.

Sea state was the third most important environmental parameter influencing flush distances. Flush distances of Common Scoter and Long-tailed Ducks decreased with increasing sea state, possibly because higher waves make it harder for the birds to spot an approaching vessel. It is also possible that birds might feel safer during higher sea states (i.e., during higher winds), as they can take off more easily with the aid of strong winds. Flush distances of Common Eiders, however, were longest during highest sea states, which may be a consequence of their higher wing loading, which makes taking off less energetically expensive during strong winds.

In addition to the environmental parameters investigated, a certain degree of variability in flush distances might be explained by molting state, as wing molt would be expected to increase energetic costs (e.g., Bridge 2004), as well as affecting wing loading and thus take-off capabilities (e.g., Guillemette and Ouellet 2005). Among the species investigated, Common Scoters show the most extensive wing molt, accompanied by a period of flightlessness (Mendel et al. 2008).

Energetic implications

This study only investigated escape reactions, which are the final and most energetically costly result of a disturbance event. However, physiologic reactions, such as an increase in heart rate (e.g., Wilson et al. 1991), will be apparent much sooner. Being alarmed and escaping leads to both a loss of foraging time and increased energy expenditure. Furthermore, foraging is less efficient during, and for a considerable time after, disturbance (e.g., Burger and Gochfeld 1998, Stolen 2003, Merkel et al. 2009). Consequently, birds will have to forage at higher rates to make up for their increased overall energy expenditure. A reduction in feeding time can even result in higher direct mortality (e.g., Mikola et al. 1994). Shorebirds are able to compensate for time lost through disturbances by increasing feeding rates or by increasing total feeding time (Swennen et al. 1989, Urfi et al. 1996). However, feeding time in sea ducks and

TABLE 4. Extended.

Velvet Scoter	Red- and Black-throated Diver
moderate to high	very high†‡
low	low†‡
species not found in shipping lanes	lower densities in shipping lanes
moderate to high	not measured
molting aggregation during summer in Baltic Sea¶ when recreational traffic is intense†	large flight distances may be result of low wing loadings; we observed only escape by flight, not by diving

loons is limited by diving time and the time needed for crushing and digesting shellfish (Guillemette et al. 1992), and therefore cannot be limitlessly extended.

Conclusions and recommendations for management

In conclusion, our findings show that the species investigated showed strong behavioral responses and, possibly more importantly, altered distribution patterns in response to ship traffic. Different species responded differently, as summarized in Table 4.

Disturbance by passing ships does not cause direct seabird mortality, in contrast to other anthropogenic activities such as gill net fishing (e.g., Dagys and Zydelis 2002). However, our data indicate that the current level of shipping intensity can already cause a (temporal) loss of foraging time and resting habitat in seabirds sensitive to human disturbance. Our results are mainly based on effects of merchant traffic, which mainly uses defined shipping routes. In contrast, recreational shipping and fishing vessels are more unpredictable in terms of speed and course and thus their disturbance potential for birds may be enhanced (for case studies, see Burger 1998, Rodgers and Schwikert 2003). The positions of most of these vessels are unrecorded or only partially recorded automatically (Vessel Monitoring System for larger fishing vessels), and information on their overlap with seabird distributions and possible habitats is therefore missing. Reliable data on the distribution of smaller vessels are urgently needed to enable their effects to be studied. The present study suggests that habituation (at least in some species) is only possible in response to constant channeled ship traffic. It is therefore recommended that spatial planning should aim to channel ship traffic wherever possible to avoid habitat fragmentation and to allow for habituation. Further studies are needed to measure the flush distances of other species (such as loons and alcids) and during different seasons (including molting), as well as to relate the behavioral responses to differences in engine noise and sailing speed (e.g., Burger 1998, Rodgers and Schwikert 2003). On this basis, the next step should be an assessment of different scenarios involving spatially changed and enhanced ship traffic, in order to quantify possible habitat loss and to elucidate the energetic consequences.

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The influence of habitat and landscape on small mammals in Estonian coastal wetlands

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Abstract. We investigated the influence of habitat type and landscape composition on small mammal relative abundance and diversity in coastal wetlands in western Estonia. Seventy live-trap lines in eight representative habitats, across six wetlands revealed seven species. The most diverse habitats were reed bed and scrub woodland, whereas tall grasslands had the highest biomass. Short grass was low in species richness and relative abundance. Small mammal relative abundance, richness, and biomass were positively correlated with tall grass and mosaic habitats and negatively correlated with lower shore habitats and bare ground. Our study indicates potential nature conservation conflicts between small mammals and other biodiversity priorities in wetlands.

Key words: diversity, Estonia, habitat, landscape, small mammals, wet grasslands.

INTRODUCTION

Wetlands are of international conservation importance for the biological diversity they support, including wet grasslands, which were historically managed for low-intensity agricultural production using extensive grazing or hay-cutting (Joyce & Wade, 1998). Flood alleviation, drainage, and agricultural intensification have reduced wetland extent and quality, while abandonment of management threatens wetland biodiversity (Strijker, 2005) through reed (*Phragmites australis*) and scrub encroachment (Leibak & Lutsar, 1996; Joyce & Wade, 1998; Luhamaa et al., 2001; Burnside et al., 2007). Therefore initiatives to reinstate appropriate management to wetlands in Europe and North America have been developed. Wetlands comprise more than 30% of Estonia (Paal, 1998) but coastal wetlands have declined from an estimated 28 750 ha in the 1950s to 8000 ha in 2000 (Luhamaa et al., 2001; Kuresoo & Mägi, 2004), many being abandoned due to political and economic changes (Unwin, 1997).

Wetlands are important habitats for small mammals (Bowland & Perrin, 1993; Krištofík, 2001) but research into factors affecting their distribution and relative abundance within wetland landscapes is relatively scanty. Small mammals play a fundamental role in ecosystem functioning as they constitute the prey base for many predators (Schmidt et al., 2002) and may potentially influence vegetation

composition via selective foraging (Brown & Heske, 1990). Generally, habitats with increased structural heterogeneity positively influence small mammal abundance and richness (Ecke et al., 2002). Mammal diversity tends to be lower in open habitats, where cover providing food and resources (Silva et al., 2005) is reduced, leading to lower fecundity (Grant et al., 1982) as well as increased predation risk (Kotler, 1997; Andreassen & Ims, 1998). Changes in landscape use and management can lead to a reduction in suitable habitats for small mammals (Raoul et al., 2001; Van Apeldoorn et al., 1992), which is compounded by habitat fragmentation causing isolation of populations (Kozakiewicz, 1993; Fitzgibbon, 1997), and consequentially gene flow (Witt & Huntly, 2001). Animal movement and dispersal are affected by vegetation structure, habitat type, landscape composition, and connectivity (Szacki et al., 1993; Fitzgibbon, 1997; Mazerolle & Villard, 1999). Hence landscape mosaic and characteristics, as well as habitat type availability, are potentially influential in determining species presence and persistence (Kupfer et al., 2006).

In this study we investigated the impact of habitat type and landscape composition on small mammal relative abundance and diversity in coastal wetlands in western Estonia. This region has some of the most biologically rich wetlands in Europe supporting internationally important numbers of birds and rare plants (Paal, 1998; Puurmann & Ratas, 1998; Luhamaa et al., 2001; Rannap et al., 2004), although comparatively little is known about their small mammal communities and the extent to which they utilize different habitats within the wetland landscape. The analysis of scale is important within landscape ecological studies as scale relates to the spatial and temporal dimensions of a landscape or habitat system (Farina, 1998). Forman (1997) defines the landscape as a mosaic where the mix of land use or ecosystems occurs over kilometre-wide areas, and habitats are local in scale, relatively homogeneous and distinct by their boundary. We surveyed small mammals in six different wetland sites with representative management intensities, habitat types, and landscape composition. Our objectives were to (a) assess species abundance, composition, and structure of small mammal assemblages in representative wetland habitats; (b) assess landscape characteristics of the wetland sites in relation to small mammal assemblages; (c) determine which habitat features locally influence mammal abundance at the species level. Results will enable better integration of the requirements of small mammals into wetland conservation and management, such as agri-environment schemes and restoration initiatives.

MATERIALS AND METHODS

Study area and site selection

Western Estonia is characterized by limestone geology and a flat topography, with an average altitude of 50 m a.s.l. The climate is continental-temperate; temperatures can reach 30 °C in summer and fall below −5 °C in winter with an annual precipitation of approximately 550 mm (Peterson, 1994; Puurmann & Ratas, 1998).

The study was conducted in the summers of 2002 and 2003 at six coastal wetlands in west Estonia (Fig. 1). Four sites were located in Vormsi Island landscape reserve, a protected area of over 10 km^2 established in 2000. Two of these sites (Sviby (58°58′31.9″ N, 23°17′42.6″ E) and Rumpo Peninsula (58°57′47.4″ N, 23°16′32.5″ E)) had recent (<5 years) re-establishment of management in the form of extensive sheep grazing and periodic hay cutting, while the other two sites (Hullo Bay (58°59′5.93″ N, 23°13′37.1″ E) and Hosby (58°58′49.0″ N, 23°21′24.6″ E)) had been abandoned and were not managed. The final two wetlands were situated on the mainland and had been managed for at least 5 years with cattle grazing at <1 livestock unit per hectare. These were Keemu (58°43′31.1″ N, 23°40′22.8″ E) in Matsalu National Park and Tahu (58°58′57.8″ N, 23°34′3.87″ E) in Silma National Nature Reserve. For further site descriptions see Burnside et al. (2007).

Aerial photographs and GIS maps were used to select sampling locations of representative habitats within the wetland sites based on vegetation composition, mapped at a resolution of 10 m × 10 m (Burnside et al., 2007). Wetland habitat types used for small mammal sampling were reed beds, short grass, tall grass, and scrub and developing woodland. Neighbouring habitats were also sampled to enable comparisons: coniferous woodlands, deciduous woodlands, juniper scrub,

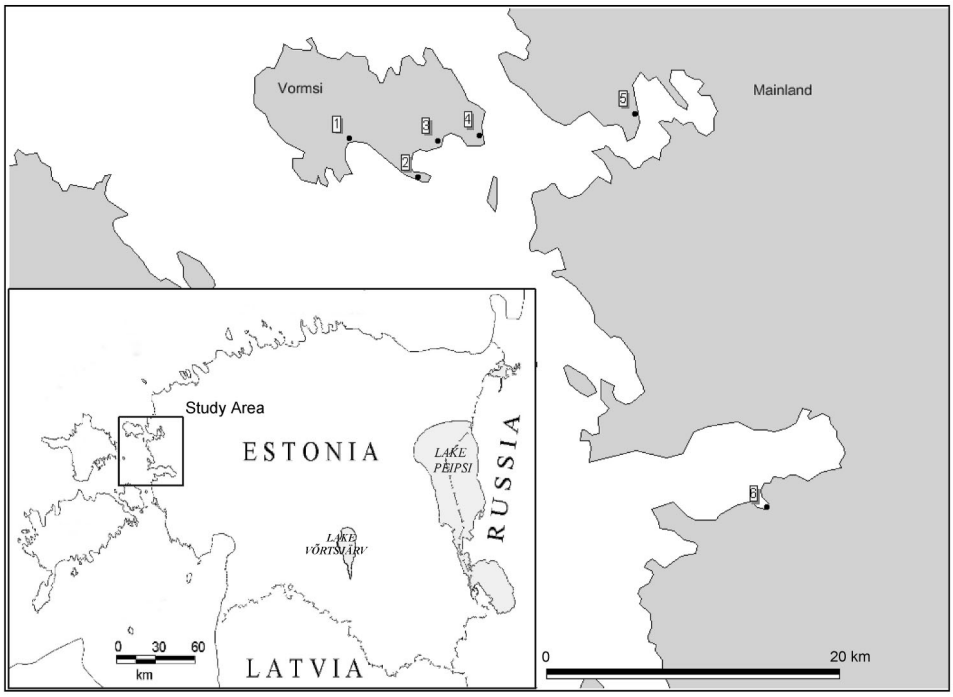


Fig. 1. Map showing locations of the six coastal wetland study sites in west Estonia. 1 = Hullo Bay; 2 = Rumpo Peninsula; 3 = Sviby; 4 = Hosby; 5 = Tahu, Silma; and 6 = Keemu, Matsalu.

agricultural field margins, and hedgerows. Trap-line locations within a site were restricted by the habitat types available, a patch size of greater than 100 m to fit a trap line within a single habitat type and a minimum distance of 100 m between samples. The aim was to maximize the diversity of wetland habitats sampled within a site irrespective of contribution to the overall wetland area. Wetland areas were also prioritized over non-wetland areas. Where the same habitat type was sampled within a site a minimum distance of 300 m was applied to aid independence. All these contributory factors led to differential sampling effort between habitats. To investigate annual variation and avoid temporal bias in communities, 23 locations were replicated annually, 15 at Sviby and the Rumpo Peninsula and eight at Matsalu. In total 70 sampling occasions were undertaken, but with 23 sites being repeated annually and averaged across the two years, this resulted in 47 locations with data.

Vegetation and habitat assessment

At each mammal sampling location vegetation structure, composition and abundance (using the DAFOR scale, Kent & Coker, 1992) were assessed using 11 variables selected considering their importance in determining small mammal communities (see Grant et al., 1982; Chętnicki & Mazurkiewicz, 1994; Ecke et al., 2002). For woodlands and hedgerows, habitat structure was separated into three distinct vegetation layers: field layer (<1 m), understorey (≥1 <5 m) and canopy (≥5 m). The percentage cover of tree and shrub canopy was recorded in four random 10 m × 10 m plots. Understorey and field layer structure and composition were determined from nine habitat variables recorded in four random 2 m × 2 m quadrats: grass cover and average height in centimetres, forb cover and average height in centimetres, reed cover and average height in centimetres, bare ground cover, litter cover, and litter depth.

Small mammal trapping

Multiple survey methods for small mammals were initially piloted, namely field sign counts (Wilkinson et al., 2004), bait tubes (Churchfield et al., 2000), and pitfall traps (Francl et al., 2002). Due to either poor returns or impracticality (i.e. non-detection, flooding of pitfall buckets), only live-trapping was eventually employed across all study areas. Small mammals were surveyed between July and August to maximize trapping yields (Masing, 1987) and because low temperatures and snow cover outside the summer period can render live-trapping unfeasible due to logistics and animal welfare. As some small mammal species have cyclic populations (Sundell et al., 2004), sampling over two years reduced temporal bias. A mammal ‘sample’ comprised three consecutive nights trapping with 10 Sherman live-traps (23 cm × 9 cm × 8 cm, H. B. Sherman Traps Inc., Orlando, Florida). The minimum number of trap nights was determined by plotting species accumulation curves from six trap nights, which showed that three nights trapping revealed the

same species as six. Rarefaction curves were also plotted for all habitats (Gotelli & Colwell, 2001). Sherman traps were used because they have been shown to attain higher species richness than pitfalls or snap-traps (Francl et al., 2002). Traps were arranged along a line transect with one trap per station spaced at 10 m intervals. Trap lines can be a better rapid method of assessing community diversity than grid trapping (Read et al., 1988). Traps were baited with a mixture of oats, peanut butter, carrots, and apple and checked twice daily at first light and again at dusk, when they were re-baited. Captures were identified to species, individually marked with a unique hair clip (Gurnell & Flowerdew, 2006), weighed, and external evidence of sexual status recorded to determine age category. All individuals were then released at their point of capture.

GIS and landscape assessment

Habitats GIS (Arc GIS 8.x ESRI) was used to quantify landscape structure at each study site. The habitat type and patch characteristics were assessed along with the surrounding landscape mosaic within each 50 m, 100 m, 150 m, and 200 m 'buffer' zone from the central GPS point of a trap line. Each increase in buffer zones included the previous area and, although not independent, gave a more realistic description of habitat use by small mammals. Each buffer zone was converted to a raster grid and the proportion of each habitat within this was calculated. Buffer sizes were selected as appropriate scales at which the landscape matrix may influence small mammal movement and dispersal (MacDonald & Barrett, 1993). The habitat patch types recorded in the buffer zones included lower shore, club-rush beds, and 'mosaic', which is defined as combinations of different vegetation types (see Burnside et al., 2007). A 'landscape mosaic' describes the spatial heterogeneity present in the landscape (Forman, 1997). FRAGSTATS 2.0 (McGarigal & Marks, 1994) was used to quantify habitat and landscape characteristics, which included total area (ha), number of patches and mean patch size (ha), edge length (m), nearest-neighbour distance (m), shape index (SI), Shannon diversity (H'), and evenness (E) (Waite, 2000).

Statistical analyses

Relative abundance was calculated as Catch Per Unit Effort (CPUE), excluding recaptures, with 0.5 trap nights removed per falsely triggered trap (Beauvais & Buskirk, 1999). Average relative biomass was calculated as total body weight trapped per 100 trap nights. Small mammal diversity was calculated using Simpson's (1-D') index (Waite, 2000). To compare habitat types the average of two years trapping was calculated for repeated sites, giving 47 samples in total. For landscape analysis only data from 2003 were used. All data were checked for normality prior to analysis. Where data were non-normal non-parametric tests were applied. Spearman rank correlation was used to relate small mammal species

relative abundance to habitat and landscape characteristics. The 11 habitat variables were checked for inter-correlation and consequently reduced to 6 variables (with Bonferroni correction $p < 0.003$), which were selected based on their likely ecological importance. As species are mainly ground dwelling percentage cover was selected over height when the two variables were inter-correlated. SPSS version 12 was used for analysis (Norusis, 1998). To further investigate patterns in communities and the influence of environmental parameters a Canonical Correspondence Analysis (CCA) was run using PcOrd (Version 4).

RESULTS

Species–habitat associations

A total of 2208 trap nights resulted in 381 captures (overall CPUE = 17%), revealing 286 individuals of seven different species: 74 yellow necked mice (*Apodemus flavicollis*), 66 field voles (*Microtus agrestis*), 52 striped field mice (*Apodemus agrarius*), 47 bank voles (*Myodes glareolus*), 44 common shrews (*Sorex araneus*), 2 harvest mice (*Micromys minutus*), and 1 water vole (*Arvicola terrestris*). No animals were found moving between trap lines, therefore they were assumed to be spatially independent. Use of all habitat data revealed no significant difference in the overall relative abundance and species richness, or individual species abundance, between samples taken on the Vormsi Island and mainland sites (*t*-test, $p > 0.05$ for relative abundance and richness; Mann–Whitney U, $p > 0.05$, for all other). In addition there was no significant difference between mainland and island individual species abundance in reed bed, short grass, or tall grass habitats (Mann–Whitney U, $p > 0.05$). Only these habitats had sufficient samples to enable comparisons. As there was no difference between mainland and island samples, data were pooled. Species habitat associations, richness, biomass, and diversity index data are summarized in Table 1. Rarefaction curves showed a species richness plateau in the majority of habitats. Too few individuals were trapped in coniferous woodland and field margin to reveal a plateau and so further trapping in these habitats may reveal more species. *Apodemus flavicollis* was the most widespread species, found in 89% of the samples and all habitats, with greatest relative abundance in deciduous woodland and hedgerows. *Sorex araneus*, *A. agrarius*, *M. agrestis*, and *M. glareolus* were also present in a wide range (67%) of samples and at least six habitat types. *Apodemus agrarius* appeared to be most common in reed bed and tall grass habitats. The relative abundance of *M. agrestis* was greatest in tall grass, whereas *M. glareolus* was found most abundantly in deciduous woodland and hedgerows. *Micromys minutus* was only trapped in two locations, a reed bed and scrub woodland, and a solitary *A. terrestris* was trapped in a hedgerow. There was no significant difference in the overall small mammal relative abundance between 2002 and 2003 (Mann–Whitney U, U = 261, $n = 23$, $p = 0.937$); however, the community composition differed between the two years (Chi-squared test, $X^2 = 678$, d.f. = 5, $p < 0.05$).

Table 1. Summary of small mammal trapping results for each habitat. RB = reed bed, SG = short grass, TG = tall grass, SW = scrub and developing woodland, CW = coniferous woodland, DW = deciduous woodland, H = hedgerow, and M = field margin. Species data are shown as average CPUE with associated standard error in italics

	Habitat															
	RB		SG		TG		SW		CW		DW		H		M	
No. of samples (Total 70)	15		19		10		5		4		8		4		3	
Species richness (Total 7)	6		4		4		6		3		4		5		4	
Average biomass, g	328.7	*99.0*	51.0	*18.3*	439	*116.8*	199.9	*84.0*	90.8	*66.9*	610.8	*229.8*	745.2	*371.2*	451.1	*322.4*
Sorex araneus	3.28	*1.46*	1.01	*0.72*	4.00	*2.37*	0.50	*0.50*	0		0.21	*0.21*	7.50	*6.44*	1.11	*1.11*
Apodemus flavicollis	2.11	*1.29*	1.05	*0.63*	1.67	*0.90*	1.33	*0.82*	0.83	*0.83*	10.71	*4.65*	14.71	*10.92*	8.89	*8.89*
Apodemus agrarius	6.11	*2.19*	0.35	*0.24*	4.67	*1.81*	3.33	*2.58*	0		0.42	*0.42*	3.33	*2.36*	0	
Micromys minutus	0.22	*0.22*	0		0		0.67	*0.67*	0		0		0		0	
Microtus agrestis	4.67	*1.95*	0.88	*0.43*	11.33	*3.38*	1.33	*1.33*	0.83	*0.83*	0		0		3.33	*1.93*
Myodes glareolus	0.44	*0.44*	0		0		2	*2.00*	1.67	*1.67*	12.05	*6.05*	5.83	*3.44*	3.33	*3.33*
Arvicola terrestris	0		0		0		0		0		0		0.83	*0.83*	0	
Simpson's diversity (1-*D*′)	0.74		0.72		0.64		0.77		0.62		0.53		0.70		0.63	

Out of the 70 samples taken 22 revealed no captures, including both samples made from the juniper scrub. Hedgerow, field margin, and tall grass trapping sessions were always successful, whereas coniferous woodland and short grass had ≥50% of samples resulting in no captures. Highest CPUE was recorded in reed bed (57%), hedgerow (57%), and deciduous woodland (70%). Overall relative abundance and species richness significantly differed between habitat types (ANOVA, $F = 2.54$, d.f. = 8, $p < 0.05$; $F = 2.18$, d.f. = 8, $p = 0.05$, respectively). However, only *M. agrestis*, *M. glareolus*, and *A. terrestris* showed significant difference in CPUE between habitat types (Kruskal–Wallis, $X^2 = 22.09$, d.f. = 8, $p < 0.01$; $X^2 = 18.52$, d.f. = 8, $p < 0.05$; $X^2 = 22.50$, d.f. = 8, $p < 0.01$, respectively). Average relative biomass also showed a statistically significant difference between habitat types (Kruskal–Wallis, $X^2 = 20.45$, d.f. = 8, $p < 0.01$) with the highest average biomass recorded in hedgerows followed by deciduous woodland. There was no significant difference in diversity between habitats. No habitat had all seven species present. Highest species richness and diversity index values were recorded in reed bed and scrub woodland. Wetland habitats had greater average species richness and average diversity, but lower average abundance compared to neighbouring habitats.

Table 2 shows the relative age and sex composition, percentage breeding, and recapture rates of animals trapped within four different wetland habitats. Other habitats were excluded as they were not deemed 'wetland'. All habitats had similar sex ratios. Short grass had the highest proportions of sub-adults and non-breeding individuals as well as the lowest recapture rates. Reed bed had the highest proportion of adult females breeding, whereas scrub and developing woodland, and tall grass, had high proportions of breeding males. Data were insufficient to allow comparisons of individual species between habitats (i.e. no species had >10 individuals in all four habitats).

Table 2. The percentage of adults, females, breeding individuals, and re-capture rates from small mammal captures in four different wetland habitats. RB = reed bed, SG = short grass, TG = tall grass, SW = scrub and developing woodland

Habitat		% Adults	% Females	% Breeding	% Recaptures
RB	Male	57	44	48	19.7
	Female	78		78	
	Total	66		61	
SG	Male	50	54	20	11.3
	Female	57		42	
	Total	54		33	
TG	Male	81	60	75	20.0
	Female	58		54	
	Total	68		63	
SW	Male	83	50	83	20.6
	Female	50		33	
	Total	67		62	

Effects of habitat and landscape characteristics

Habitat variables were tested for correlations with small mammal species relative abundance (Table 3). As vegetation cover differed annually, each of the 70 samples was kept separate. Due to the low number of captures *A. terrestris* and *M. minutus* were excluded from the analysis. The relative abundance of *A. flavicollis* and *M. glareolus* was significantly negatively correlated with grass cover and significantly positively related to tree density. *Apodemus agrarius* was significantly positively correlated with litter depth and reed cover, whereas *M. agrestis* was significantly positively correlated with grass cover and showed a significant negative association with bare ground and tree density. There were also three significant relationships between species, with the relative abundance of *M. agrestis* and *A. agrarius*, *M. glareolus* and *A. flavicollis*, and *A. terrestris* and *S. araneus* all positively correlated (Table 3). The ordinations of environmental variables in the first two canonical axes were plotted using bi-plot scores from CCA (Fig. 2).

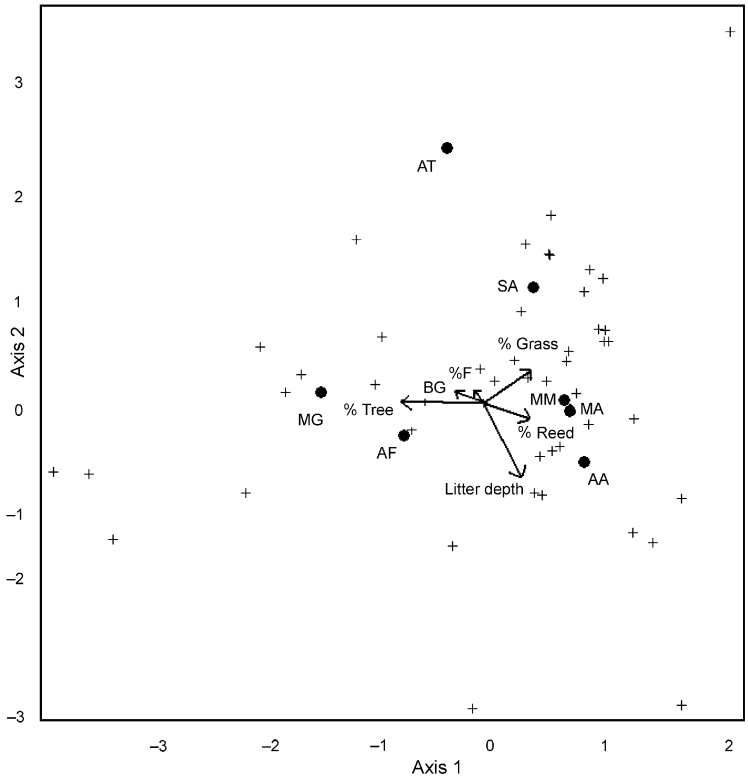


Fig. 2. Bi-plot and LC scores from Canonical Correspondence Analysis plotted for species, sample, and habitat variables (No. of samples = 47) in the first two canonical axes. Here ● represents species, + represents sites, and arrows represent environmental variables. AF = *Apodemus flavicollis*, SA = *Sorex araneus*, AA = *Apodemus agrarius*, MA = *Microtus agrestis*, MG = *Myodes glareolus*, MM = *Micomys minutus*, and AT = *Arvicola terrestris*. % is percentage cover, BG = bare ground, F = forbs and herbs.

Table 3. Spearman rank correlation coefficient between the relative abundance of small mammal species and key habitat characteristics. SA = *Sorex araneus*, AF = *Apodemus flavicollis*, AA = *Apodemus agrarius*, MA = *Microtus agrestis*, MG = *Myodes glareolus*, AT = *Arvicola terrestris*, and MM = *Micromys minutus*. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$

Species	Habitat variables						Species					
	% Forbs	% Grass	Bare ground	Litter depth	Tree density	% Reed	SA	AF	AA	MA	MG	AT
SA	−0.108	0.150	−0.044	−0.161	−0.135	0.104						
AF	0.037	−0.272*	0.127	0.014	0.316**	−0.165	0.109					
AA	−0.102	−0.118	−0.221	0.486***	−0.124	0.400***	0.122	0.206				
MA	−0.102	0.304*	−0.243*	0.087	−0.306**	0.147	0.162	−0.113	0.366**			
MG	0.021	−0.281*	0.209	−0.113	0.461***	−0.174	0.082	0.519***	−0.035	−0.212		
AT	0.200	−0.080	0.003	−0.116	0.161	−0.075	0.268*	0.174	−0.072	−0.080	−0.052	
MM	−0.132	0.093	−0.079	0.189	0.013	0.098	−0.100	0.031	0.136	0.116	0.164	−0.021

The first canonical axis correlated most highly with % tree cover to % reed cover. However, this axis accounted for only 20.3% of the variance (eigenvalue = 0.424, $R = 0.76$). The combined first, second, and third axes only accounted for 36.3% of the variance. A Monte Carlo permutation test showed that the first canonical axis was significantly correlated to species distribution patterns ($p = 0.02$).

Only 29 trap locations had sufficient spatial mapping extending to 200 m zones to enable landscape analysis. The presence of small mammals was not related to the size of the wetland habitat patch in which trapping took place, as the extent of tall grass, reed bed, short grass, or scrub and developing woodland surrounding the trap lines was not correlated with mammal relative abundance, richness, or biomass. However, small mammal parameters tended to be significantly positively correlated with landscapes containing tall grass and mosaic habitats, and significantly negatively correlated with larger areas of short grass, club rush swamp, lower shore, and bare ground (Table 4). Mammal species richness and diversity were particularly sensitive to landscape composition, being significantly correlated to the extent of each habitat type at nearly all spatial scales (Table 4).

Table 4. Correlation coefficients showing the relationship between small mammal parameters and landscape composition (% cover) at four different zone sizes ($n = 29$). R.A. = relative abundance, Diversity = Simpson's diversity (1-D'). TG = tall grass, LS = lower shore, SG = short grass, BG = bare ground, CB = club rush swamp. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$. Spearman rank correlation was used, gaps represent no significant correlation

Zone size, m	Habitat	Small mammal parameter			
		R.A.	Species richness	Diversity	Biomass
50	Mosaic		0.428**	0.428**	
	TG	0.660***	0.632***	0.544**	0.652***
	LS		–0.384**	–0.458*	0.404**
	SG		–0.380**		
100	Mosaic	0.663***	0.708***	0.657***	0.643***
	TG				0.399*
	CB	–0.574**	–0.599**	–0.508**	–0.520**
	BG		0.441*	0.502**	
150	Mosaic	0.562**	0.692***	0.606***	0.550**
	TG	0.643***	0.621***	0.629***	0.556**
	CB	–0.461*	–0.509**		–0.419*
	LS	–0.529**	–0.592**	–0.657***	–0.516**
	BG	–0.442*	–0.447*		–0.441*
200	Mosaic	0.538**	0.678***	0.607***	0.552**
	TG	0.610***	0.623***	0.620***	0.550**
	CB	–0.506**	–0.567**	–0.397*	–0.462*
	LS	–0.573**	–0.649***	–0.657***	–0.556**
	SG	–0.458*	–0.464*	–0.380*	–0.394*
	BG	–0.503**	–0.509**		–0.502**

All other recorded landscape variables ($n = 20$) did not show any correlation with mammal relative abundance, richness, or biomass (Pearson product correlation $p > 0.05$ for all).

DISCUSSION

Although seven representative species (e.g. see Masing, 1987) were recorded, rarer species may have been under-represented due to survey length, timing, or method. In comparison to data collected by Masing (1987) in Matsalu, we did not capture common voles (*Microtus arvalis*) or Northern birch mice (*Sicista betulina*). However, Masing's study encompassed agricultural and urban habitats as well as coastal. Sherman traps can bias for species >10 g (Francl et al., 2002), this bias may have affected *S. betulina*, which has a body mass range of 5–13 g (MacDonald & Barrett, 1993). Additionally water shrews (*Neomys fodiens*) were expected to be found, but were not captured. As trap success of water shrews tends to be low, an alternative method to effectively survey them is to set bait tubes and identify faecal remains (Churchfield et al., 2000). It is therefore recommended that in future surveys a variety of methods be employed to detect a wider range of species. While standardized trap lines provide a rapid assessment of presence and relative abundance, they do not yield density estimates (Gurnell & Flowerdew, 2006). Whilst the design of this survey enabled comparisons across a range of habitats, variation in capture rates suggests modifications in future surveys are needed to account for density variations. Also, although 70 transect lines were sampled in six different wetland sites with over 2200 trap nights, the number of small mammal captures was low (381, 17% CPUE) and high variability was found within habitats. Whilst the value of this study is partially derived from the paucity of information on small mammals in wetland landscapes, particularly in Estonia, greater capture effort and returns would clearly help verify findings. Further studies should also investigate spatial and temporal utilization of the coastal wetland landscape by small mammals to determine the relative importance of different habitats at different times of the year, as this study was limited to summer sampling. Community assemblages will vary spatially and temporally. Populations of *Microtus agrestis* in Finland have been recorded to fluctuate on a three year cycle (Huitu et al., 2003), hence the timing of the study in relation to population cycles would affect results. In addition, Huitu et al. (2003) found landscape composition to influence cycles, therefore this also needs to be taken into account. Habitat quality is an important factor determining mammal presence and abundance (Poulin et al., 2002) and may explain the large differences within samples recorded in this survey. As flood frequency and duration have direct impacts on small mammals (Jacob, 2003), water level monitoring may also help to understand temporal and spatial patterns in small mammal habitat associations.

Habitat and landscape effects

The results of this study in west Estonia support research indicating the importance of wetlands for small mammals (e.g. Krištofik, 2001). For example, reed beds had the highest species diversity of all habitats surveyed. Large variations in capture rates (0–90%) may have been due to reed bed quality and/or hydrological regime, which was found to determine reed bed dwelling birds in France (Poulin et al., 2002). A study of small mammals in reed stands in Slovakia resulted in 16 species, with the community dominated by *S. araneus*, *M. glareolus*, and *A. sylvaticus* (Krištofik, 2001). Reed beds in west Estonian sites were dominated by *A. agrarius* followed by *M. agrestis* and then *S. araneus*. Differences in dominance may be due to the geographic species pool (MacDonald & Barrett, 1993), shifts in habitat preferences in relation to competition, or reed age, type, or quality (Poulin et al., 2002). Reed beds supported the highest proportion of breeding female small mammals, possibly due to a combination of cover and inaccessibility providing protection from predators, and plentiful resources for both nest construction and meeting the increased metabolic demands during weaning. In a study in Austria, Haberl & Kryštufek (2003) found harvest mice (*M. minutus*) at densities of up to 93 individuals per hectare in reed beds, suggesting that this could be an important habitat for this internationally 'near threatened' species (Amori, 1996). Tall grass had the same mammal species composition as short grass but in greater relative abundance, probably because short, highly grazed or cut grass is commonly avoided by small mammals (Grant et al., 1982; Eccard et al., 2000; Tattersall et al., 2001; Schmidt et al., 2005). Short grass had the highest proportions of sub-adults and non-breeding individuals as well as the lowest recapture rates, indicating that this is less likely to be a breeding habitat and that animals captured here are mostly transitory, dispersing sub-adults or non-breeding adults. However with sufficient data, species specific analysis should be undertaken to elucidate the role of each habitat to each species. Short grass habitats seem more likely to be 'sink' as opposed to 'source' habitats for small mammals. Low recapture rates also indicate that this habitat is probably used during dispersal rather than maintaining a high resident population. Scrub and developing woodland habitats had relatively high species richness, although most species were in low relative abundance. This habitat is typically positioned on higher ground at the wetland periphery and may offer temporary refuge during flood events. Agricultural field margins and hedgerows were utilized by a range of small mammals. Hedgerows provide protection from predators, food sources, and can act as corridors for dispersal, especially when adjacent land use is lower in cover and higher in predation risk (Chętnicki & Mazurkiewicz, 1994; Tattersall et al., 2001).

Mammal relative abundance, diversity, biomass, and richness were positively correlated with the amount of adjacent mosaic and tall grass habitat within 200 m of the trapping site and negatively correlated with more open landscapes supporting habitats such as short grass, bare ground, and lower shore herbaceous vegetation.

Tall grass provides cover and protection from predators for small mammals (Tattersall et al., 2001) whereas short grass and open patches provide little protection and tend to be avoided (Eccard et al., 2000; Tattersall et al., 2001).

Implications for conservation

Management of wetlands through grazing and cutting for agriculture or conservation often leads to a reduction of those wetland habitats such as reed bed and tall grass that in this study were found to support high diversity and relative abundance of small mammals. Such management favours short grass and lower shore habitats (see Burnside et al., 2007), which contained relatively low small mammal richness and relative abundance in this study. Estonian wetlands are of international conservation concern due in part to the wetland birds that utilize them (Joyce & Wade, 1998). Many of these birds, such as waders, require medium-short grass for feeding and successful breeding (Kuresoo & Mägi, 2004) and large open areas reduce the risk of predation (Butler et al., 2005), both of which are less suitable for small mammals. At the landscape scale, management for specific conservation or agricultural targets may result in greater uniformity with less mosaic vegetation, which this study suggests would also be detrimental to small mammals. Furthermore, isolation of suitable habitat patches may prevent movement between habitats and consequently more isolated patches tend to have lower small mammal populations (Marsh et al., 2001). This illustrates a conflict of interest in nature conservation and highlights the need for a holistic approach to restoration and conservation management of wetlands, including a better understanding of the impacts of conservation management practices on different (e.g. non-target) components of the ecosystem. Such a perspective for wetland landscapes may seek to provide a mosaic of habitat types for a range of different species, although in practice reconciling the needs of different biodiversity components is difficult. For small mammals, however, this study has indicated that biodiversity management should aim to incorporate refuge habitats such as reed beds, tall grassland and scrub within the wetland landscape.

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Elupaiga tüübi ja maastiku mitmekesisuse mõju pisiimetajatele Eesti ranniku-märgaladel

Dawn M. Scott, Chris B. Joyce ja Niall G. Burnside

On uuritud elupaiga tüübi ja maastiku struktuuri mõju pisiimetajate suhtelisele arvukusele ning liigilisele mitmekesisusele Lääne-Eesti ranniku-märgaladel. Kuuelt märgalalt, kokku kaheksast biotoobist, õnnestus tabada seitsmekümne eluspüügi-lõksuliiniga seitse liiki pisiimetajaid. Liigiline mitmekesisus oli kõrgeim roostikes ja põõsastikes ning pisiimetajate biomass oli kõrgeim niitmata rohumaadel. Madalmurustel rohumaadel olid madal nii liigirohkus kui isendite hulk. Pisiimetajate arvukus, liigirohkus ja biomass seostusid positiivselt niitmata rohumaa ning mosaiikelupaiga hulgaga ümbruses ja negatiivselt vahetu mere kalda ning avatud pinnasega alade hulgaga. Uuringu tulemused viitavad võimalikule looduskaitselisele vastuolule pisiimetajate ja ülejäänud biomitmekesisuse kaitsel märgaladel.

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Traffic noise reduces foraging efficiency in wild owls

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Anthropogenic noise has been increasing globally. Laboratory experiments suggest that noise disrupts foraging behavior across a range of species, but to reveal the full impacts of noise, we must examine the impacts of noise on foraging behavior among species in the wild. Owls are widespread nocturnal top predators and use prey rustling sounds for localizing prey when hunting. We conducted field experiments to examine the effect of traffic noise on owls' ability to detect prey. Results suggest that foraging efficiency declines with increasing traffic noise levels due to acoustic masking and/or distraction and aversion to traffic noise. Moreover, we estimate that effects of traffic noise on owls' ability to detect prey reach >120 m from a road, which is larger than the distance estimated from captive studies with bats. Our study provides the first evidence that noise reduces foraging efficiency in wild animals, and highlights the possible pervasive impacts of noise.

Anthropogenic noise (hereafter "noise") is increasing globally and mounting evidence suggests that noise can negatively affect wild animals in many ways[1–3]. Of these impacts, masking from noise, where it interferes with an organism's ability to detect or discriminate biologically relevant signals, appears to be especially problematic[4–6]. Although several studies have examined impacts of masking using "quiet *versus* loud designs", to fully understand and reduce the severity of masking, quantifying wildlife responses to a range of noise exposure levels is critical[5–7].

Compromised foraging efficiency in animals, especially in acoustic predators such as owls and bats, is among the main concerns regarding impacts of novel acoustic environments created by noise[8–10]. This is because declines in foraging efficiency likely influence their distributions by altering behavior and reducing habitat suitability[11,12] and thereby may alter predator-prey interactions that have ecosystem-wide consequences[13]. Nevertheless, only a few laboratory experiments with limited sample sizes have examined noise impacts on foraging efficiency, and only in two bat species[8,11,12] and in a single owl species[14]. Thus, to clarify whether negative effects of noise on foraging efficiency in acoustic predators are widespread, we must understand the degree to which noise degrades foraging efficiency in acoustic predators in the wild[5,6].

The objective of this study was to experimentally determine the relationship between foraging efficiency of wild acoustic predators and noise levels common to many landscapes. We studied nocturnal owls because they are specialized acoustic predators, have cosmopolitan distributions, and have different audible ranges and hunting techniques than bats. We conducted novel field playback experiments using two types of sounds, traffic noise (hereafter "TN") and artificial prey rustling sound (hereafter "APRS") (Fig. 1). Playback of TN allowed us to isolate effects of noise from other confounding factors, such as habitat changes, visual disturbance of moving vehicles and lights, etc[15–17]. Because owls localize and attack prey using prey-generated rustling sounds at frequencies spanning 6–8.5 kHz[18], we digitally developed APRS (Fig. 2a) and found that owls in the wild are attracted to playback of these sounds (see supplementary Fig. S1), providing a method for quantifying prey detection among wild owls under a variety of acoustical conditions. In field experiments, we played back APRS at constant amplitude under various TN exposure levels at many locations in northern Japan, and thereby examined the effect of TN on owls' ability to detect APRS. Finally, we estimated the compromised foraging range by noise near roads. To the best of our knowledge, this is the first study to examine effects of different levels of TN on foraging efficiency in acoustic predators in the wild.

Results

We conducted 367 playback experiments in northern Japan (see supplementary Fig. S2), and recorded a total of 92 owls in 76 playback experiments (Table 1). After exclusion of owls that did not satisfy our analytical criteria

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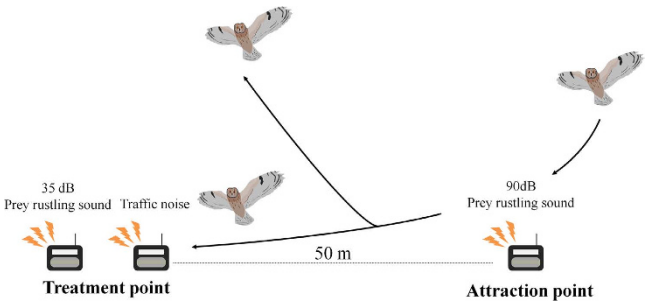


Figure 1. Schematic of the playback experimental set up.

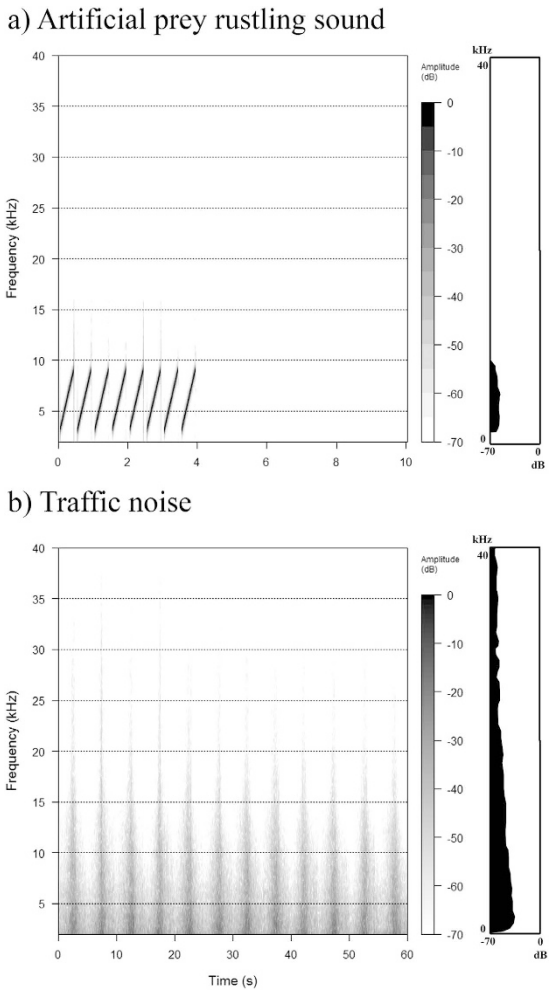


Figure 2. Spectral characters, relative amplitudes (left panel) and power spectra (right panel) of (**a**) ARPS and (**b**) TN.

	Yufutsu	Sendai	Total
Number of study plots	45	58	103
Number of experiments	210	157	367
Number of owls	21	71	92
long-eared owl	7	29	36
short-eared owl	14	39	53
ural owl	0	3	3

Table 1. Summary of field experiments.

Variables	Model rank				β	SE
	1	2	3	4		
Traffic noise level						
Distance from road	+	+			−0.30	0.01
Distance from road^2	+		+		0.00	0.00
df	4	3	3	2		
ΔAICc	0.00	115.80	199.44	295.71		
Weight	1.00	0.00	0.00	0.00		
Owls' prey detectability						
Trafic noise	+	+	+		−0.07	0.02
Species_ID		+	+			
TN X SP_ID			+			
df	4	5	6	3		
ΔAICc	0.00	2.28	4.60	17.07		
Weight	0.70	0.23	0.07	0.00		

Table 2. Results of GLM examining how TN decreases with distance from a road and GLMM examining effects of TN on owl's ability to detect prey. For GLM, we treated SPL as a response variable, and distance from a road (m) and its quadratic term as explanatory variables. For GLMM, we treated whether owls detected APRS at the treatment point as the response variable, SPL of TN, species ID and interaction of these variables as explanatory variables and plot ID and Study region (Yufutsu or Sendai) as random variables. Variables included in models are indicated with plus sign. "TN X SP_ID" indicates the interaction term between traffic noise and species ID and "Weight" refers to Akaike Weights. Parameter estimates (β) and its standard errors (SE) in the best models are also given.

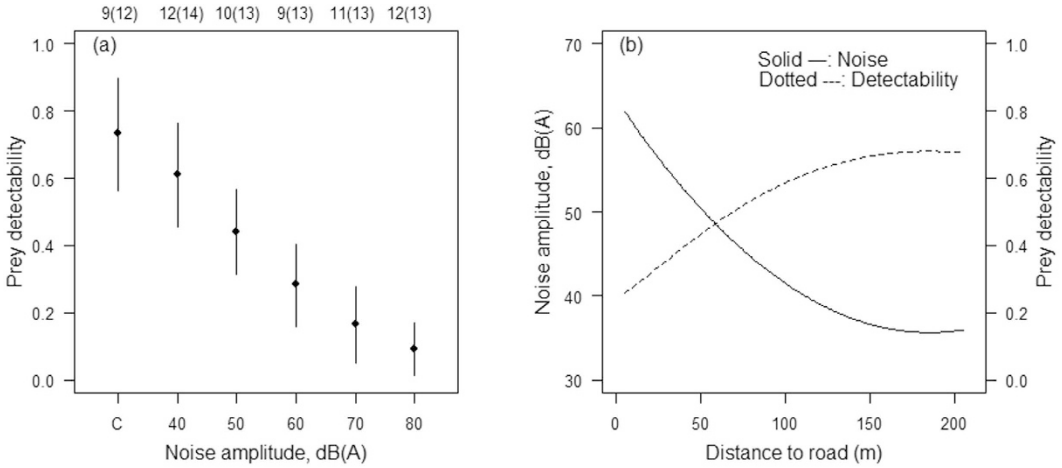


Figure 3. (**a**) Estimated owls' ability to detect prey under noise exposure levels (with 95% CI). "C" indicates control experiments. Detectability at C is estimated using average background sound level (32 dB). Top figures indicate number of experiments (number of owls analyzed). (**b**) Relationships between road distances and noise levels and owls' prey detectability. The owls' ability to detect prey was estimated based on linear regression equation presented in (**a**).

($n = 14$, see "Materials and Methods"), we analyzed 78 owls in 63 playback experiments (45 short-eared owls *Asio flammeus* and 33 long-eared owls *Asio otus*). The best model included sound pressure level (SPL) of TN and suggested owls' ability to detect prey was negatively associated with SPL of TN (Table 2, Fig. 3a). Species ID and the interaction term had no association with owls' ability to detect prey (Table 2).

In addition, to estimate relationship between road distances and owls' ability to detect prey, we also measured SPL of TN at various distances from road. Result of model selection showed SPL of TN attenuated quadratically with distance from road (Table 2, Fig. 3b), indicating impacts of traffic noise on owls' ability to detect prey has the potential to reach >120 m from a road (Fig. 3b). In other words, owls' ability to detect prey was impacted even at the lowest level of TN (40 dB[A]) and was approximately 17% lower than detections in ambient sound conditions (Fig. 3a).

Discussion

Using a novel field-based experimental approach, we show that owls' ability to detect prey is negatively impacted by increases in TN. Masking of the signal occurs when there is spectral and temporal overlap between the signal and noise[5]. Because APRS experience considerable spectral overlap by TN (Fig. 2a,b), reduction of owls' ability to detect prey could be caused by increasing acoustic masking with increasing amplitude of TN[11,12], although whether the signal was masked depends on the acoustic processing abilities of the owls and how they hear sound in noise. Additionally, it is not mutually exclusive that distraction and/or avoidance to TN play some role for explaining decreases in prey detection. For example, previous works with bats suggests that distraction/avoidance to noise had larger impact on bats' ability to detect prey than masking[8].

Owls' ability to detect prey was impacted even at the lowest level of TN (40 dB[A]) and was approximately 17% lower than that of ambient conditions (Fig. 3a). This corresponds to a distance of 120 m estimated from our model predicting noise levels from distance to the road and is twice the distance estimated for impacts on bats' prey detectability due to TN[9]. Methodological differences between laboratory and field studies could explain these differences. For example, laboratory experiments conducted in a restricted area could overestimate prey detectability of acoustic predators because in a confined laboratory setting they circle above the experimental foraging area in flight and may have more chances to detect their prey. In contrast, in the field acoustic predators typically forage in linear flight and would have fewer opportunities to detect prey sounds[11]. Alternatively, such difference may be, at least partially, due to differences of audible range and/or sensitivity to sounds between birds and bats (i.e., owls cannot detect sounds at frequencies above 15 kHz[18] while echolocating bats use sounds at frequencies up to 120 kHz)[11] and differences in prey-generated rustling sounds between the experiments.

Masking of real or artificial prey rustling sounds by traffic noise should invariably reduce foraging efficiency to some degree. However, hunting owls may be able to take advantage of directional masking release where rustling sounds and background noise propagate from different directions. Distraction, in which owls attend to traffic noise rather than rustling sounds, could also explain declines in prey detectability and could operate along side masking. However, it is also possible that distraction or compromised attention could decrease with habituation to traffic noise over time. Distinguishing among these potential mechanisms must be a next step. Additionally, it is also critical to understand whether declines in prey detection scale to responses most relevant to population persistence, such as site abandonment or impact actual foraging success, body condition and reproductive success of animals occupying noisy areas[11,19,20].

In addition, there are several differences between this study and natural conditions. First, omnidirectional TN used here differs from horizontal TN propagation from roadways that wild owls encounter in nature. Thus, future work evaluating how directional masking release changes detection of APRS is needed. Second, we used a representative TN sound recording in the experiments based on comparisons among several TN sounds. Although this isolates noise amplitude as a single factor that varied among treatment levels, it also does not reflect TN variation due to variable traffic speeds, densities and environmental conditions, indicating that future work should focus on how possible TN variation affects owls' ability to detect prey. Moreover, high frequency components of TN attenuate faster with distance from roads than lower frequency components, suggesting overestimation of the masking effects of TN playbacks at amplitudes reflective of 55, 105, 155, 205 m from the road. However, because APRS playbacks were louder than natural prey rustling sounds and APRS might be easier for owls to detect than actual prey rustling sounds with broadband energy, effects of TN on owls' prey detection may extend well-beyond our 120 m estimate.

Despite the need to parse the effects of how directional masking release, real versus artificial prey sounds and high frequency components simultaneously contribute to estimated impacts with respect to distance from roads, we provide the first evidence that noise reduces foraging efficiency in a wild predator in a natural situation. Additionally, our analysis of sound level attenuation with distance from the road suggests that declines in prey detection occur at distances twice that estimated for bats from lab studies[11], at least in our study region. Nevertheless, given our playback is representative of traffic noise propagating from other roadways (see supplementary Fig. S3), it is likely that impairment of foraging at similar distances is generalizable to other roadways. Moreover, a recently published captive study showed that experimental playback of compressor noise, which has similar power spectrum with traffic noise, negatively impacts hunting behavior of northern saw-whet owls (*Aegolius acadius*) at sound levels as low as 46 dB(A), which corresponds to approximately 800 m from compressor stations[14]. These potentially sizable footprints from energy-sector and traffic noise highlight the pervasive impacts of noise on acoustic predators because many sources of noise, including road densities, are high and increasing[4]. For example, 83% of the continental US is within 1061 m of a road[21], and globally, >25 million kilometers of new roads are anticipated by 2050[22]. Key to fully understanding noise-impacts on acoustic predators will require knowledge of how the magnitude of noise-impacts varies depending on road densities, arrangements and traffic volumes and speeds. Moreover, it is critical to understand how common prey species respond to roadways and traffic and determine whether the cumulative effects are additive, synergistic or even antagonistic, as some nocturnal small mammals appear to increase in noise exposed areas[23] and along roadways[24]. Regardless of the shape of these interactions, it is likely that wild owls and other acoustically-oriented predators will continue to be impacted by noise.

Methods

(a) Preparation of the traffic noise for playback experiments. Vehicle noise was recorded at the prefectural road #1046 in Yufutsu plain, central Hokkaido, late December 2014. The recording was conducted between 22:00 to 02:00 on a clear day when wind speeds were less than 1 m/s. We set a recorder (PCM-D100, Sony Corporation, Tokyo, Japan; frequency response ± 2 dB between 20 Hz and 45 kHz) with a sound pressure meter (Sound Level Meter TYPE 6236, ACO CO., LTD, Miyazaki, Japan) at a height of 1.5 m and 5 m distance from the road. Then, for each of 20 passing vehicles at constant speed (60 km/h), we recorded its noise and measured its

sound pressure level (SPL) as the A-weighted equivalent continuous noise level during five seconds at nearest distance to a vehicle (L_{eq} [5 s], fast response time, re. 20 μPa, A-weighting). For these, we used the A-weighted filter because this filter provides better measurement of acoustic energy relevant to birds at frequencies between 1.0 and 9.5 kHz[25], which cover entire frequency range used by hunting owls[18]. Finally, we created a 1 min exemplar of TN sound consisting of 12 vehicle pass-by events, which contained energy up to 40 kHz, but had the most energy below 10 kHz (Fig. 2b). This traffic level was found along roads in many national parks, national forests, and protected areas globally[19]. Although it is better to use different TN sounds in each playback experiment to capture potential heterogeneity in traffic noise present in different locations or times, because our primary interest is to quantify effect of amplitude alone on owls' ability to detect prey, we used this single TN sound file in all playback experiments based on comparisons of frequency spectra among several TN sounds recorded at different locations (see supplementary Fig. S3). We also created a 1 min control sound file that had no acoustic energy. In addition, to understand how sound levels attenuate with distance from the roadway, for each of 20 passing vehicles at known speed (i.e., 60 km/h), we measured its sound pressure level (SPL) as the A-weighted equivalent continuous noise level during five seconds at nearest distance to a vehicle (e.g., LAeq [5 s]) at 5, 55, 105, 155, and 205 m from the road.

(b) Preparation of the artificial prey rustling sound for playback experiments. When small-mammals walk on the ground, they produce rustling sounds which are short and contain a wide range of frequencies[18]. Owls can precisely locate these rustling sounds, especially at frequencies between 6 and 8.5 kHz[18]. Because they respond strongly to stimuli at these frequencies, we created sound files consisting of an upsweeping element of 0.4 s in duration spanning 3.0–9.0 kHz separated by 0.1 s (sampling rate: 192 kHz). For each file, the elements were repeated eight times, followed by 6 s with no acoustic energy (Fig. 2a). This 10 s section was then repeated six times to create a one-minute artificial prey rustling sound, which is similar in structure to rustling sounds made by actual prey[18]. All sound analyses and clip generation were conducted in Sound Forge Audio Studio 10.0 (Sony, Tokyo, Japan).

(c) Study area and field playback experiments. To make certain that we could obtain sufficient sample sizes, we selected two study areas in northern Japan where many owls overwinter. Specifically, field experiments were conducted in Yufutsu plain, central Hokkaido and in Sendai plain, northern Honshu (see Supplementary Fig. S2). Both landscapes are predominantly agricultural fields and semi-natural grasslands (see Supplementary Fig. S2), providing suitable environments for our target study species. We established 103 playback experimental plots in these two areas (45 in Yufutsu plain and 58 in Sendai plain, northern Honshu) (see Supplementary Fig. S2). In the study area, an individual short-eared owl territory size was estimated to be approximately 5 ha (M. Senzaki, personal observations), which nearly equals an area with 130 m radius. Thus, to prevent double sampling, adjacent plots were spaced by $>$500 m. In addition, we did not establish plots in areas with tall trees or streetlights to prevent potential effects of these factors on sound propagation or owls' behaviors respectively. Playback experiments were conducted at least once in each plot between 1700–0500 h on both clear and cloudy nights, when wind speeds were $<$2 m/s, from December 2014 to March 2015, which corresponded with owls' wintering periods. The average number of playback experiments at each plot ($\pm$SD) was 3.56 ± 1.23. When owls were sampled in a plot, we did not conduct any additional playback experiments in the same plot three or more days to minimize effects of habituation.

A plot consisted of an attraction and treatment point spaced 50 m apart (Fig. 1) with one and two speakers (PDX-B11: Yamaha, Hamamatsu, Japan; frequency response $\pm$ 10 dB between 55–20 kHz) connected with players (WALKMAN NW-E080, Sony Corporation, Tokyo, Japan), respectively. Although traffic noise propagates horizontally across the landscape, and mimicking directional propagation can be carefully controlled in laboratory conditions[11,12], we set all speakers on the ground facing upwards to ensure omnidirectional propagation of attraction and treatment point sounds across the landscape. This ensures fairly equal amplitudes of playback sounds in all directions, which was important when owls could approach from any direction. On nights when background SPL $\leq$ 35 dB(LAeq[1 min]), we broadcasted TN or the silent sound file with no acoustic energy (hereafter "control sound") from one speaker at the treatment point until the end of the experiment. Amplitude of TN was randomly chosen to be approximately 40, 50, 60, 70, or 80 dB(LAeq [5 s]) at 1.5 m height above the speaker, representing sound levels measured at different distances from a roadway. After 1 min of TN broadcast at the treatment point, we first broadcast APRS at 90 max dB(A) at a height of 1.5 m above the attraction point speaker for 1 min to attract owls from the larger surrounding area. When the playback was finished, we immediately broadcast APRS at 35 max dB(A) at a height of 1.5 m above the second speaker at the treatment point for 1 min. Although 35 dB(A) is louder than natural prey sounds[11,12], we used the value to ensure that owls at attraction points could detect APRS at treatment points at least under control playback conditions. We tracked owls attracted to the attraction point and determined whether they could subsequently detect APRS at the treatment point. Owls that actively entered the range within 10 m from the attraction/treatment point (e.g., owls hovering and/or flying circular over the speaker) were determined to detect APRS in each point. When we observed attacks and/or chases between attracted owls and/or when we could not determine whether owls in attraction points were detecting APRS in treatment points because they landed on the ground, they were not included in subsequent analyses. We also excluded experiments with no owls detected from any analysis. Because flying owls could be observed at approximately 50 m distance from an observer, observations were conducted 30 m from both attraction and treatment points using a night scope (ATN Night Spirit XT, California, USA) and binoculars (MONARCH 8 $\times$ 42, NIKON CORPORATION, Tokyo, Japan).

(d) Data analysis. We used Generalized Linear Model (GLM) with Gaussian error to examine how TN decreases with distance from a road. We treated SPL as a response variable, and distance from a road (m) and its quadratic term (m^2) as explanatory variables.

We examined effects of TN on owl's ability to detect prey using Generalized Linear Mixed Model (GLMM) with Binomial error. We treated whether owls detected APRS at the treatment point as the response variable, SPL of TN, or ambient SPL measured prior to the start of control trials, species ID (long- or short-eared owl) and the interaction of these variables as explanatory variables. Plot ID and Study region (Yufutsu or Sendai) were treated as random variables. Although identifying whether the same individuals were recorded within a specific plot was difficult due to low light levels, treating plot ID as a random effect can account for possible repeated sampling of the same individuals. For experiments with control sound, SPL measured before the start of the experiments was used. We constructed models for the combinations of all possible covariates, ranked them by Akaike's information criterion for small sample size (AICc), and considered covariates in the best model as meaningful predictors. These analyses were conducted using "lme4" (v. 1.1–5)[26] and "MuMIn"(v. 1.9.13)[27] with R software (v. 2.15.3)[28].

(e) Ethical statement. All experiments were performed in accordance with relevant guidelines and regulations. All experimental protocols were approved by the Japanese Ministry of the Environment.

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Author Contributions

M.S. designed the study, carried out the field experiments and drafted the manuscript. Y.Y. participated in the design of the study and helped draft the manuscript. C.D.F. and F.N. helped draft the manuscript. All authors gave final approval for publication.

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The diet of the Marsh Harrier *Circus aeruginosus* breeding on the isle of Hailuoto compared to other raptors in northern Finland

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The Marsh Harrier *Circus aeruginosus* population has expanded rapidly during the last decades in northern Finland. We studied the diet of Marsh Harrier during the breeding seasons 2002–2009 on the isle of Hailuoto in the Gulf of Bothnia in Finland. We compared these results to those of other raptors in the region, and attempted to relate potential differences in the diet composition to population trends. Birds accounted for 73–93% of prey numbers in the Marsh Harrier during the breeding season. Passerine birds were the most numerous during the nestling (63%) and fledging periods (21%), while ducks and gulls/terns had the highest biomass (29% and 12%, respectively). Muskrat (*Ondatra zibethica*) was the most important prey among mammals by number and weight (15% and 18%, respectively). Compositional analysis showed that the Marsh Harrier depredated water and shore birds opportunistically. Based on multidimensional scaling, the food niche of the Marsh Harrier was distinctive from those of five other raptor species. Analysis of diet breadth and annual change in population size suggested that Marsh Harrier diet breadth was wider and its annual population increase among the fastest, compared to other raptor species. Annual population change of raptors was positive among species that were rare in the beginning of monitoring, i.e., in the early 1970s, but was negative among species that were common at that time. No significant relationships were observed between diet breadth and population growth among the studied species.



1. Introduction

Concerns about the future of many raptor populations have led to special monitoring programs in many countries (Kovacs *et al.* 2008). These programs have detected both negative and positive long-term trends (Saurola 2008). Reasons for these trends may be changes in the availability of crucial resources, such as food and nesting sites (Korpimäki & Norrdahl 1991), competition (Hakkarainen *et al.* 2004), habitat quality (Widén 1997), environmental pollutants (Helander *et al.* 2002) and human persecution (Newton 1979). Some of these factors, especially pollution and human persecution, caused populations of some species to collapse during the early 20th century until the 1970s. But due to significant improvements in these factors the populations have often rapidly recovered (e.g., Stjernberg 2003).

Food is obviously a fundamental resource for

all organisms to survive and reproduce. Raptors are an ideal group for studying the connection between food (prey) availability and population dynamics due to the relatively easy determination of diet and often reliable estimation of prey abundance by counting and trapping prey species (Korpimäki & Norrdahl 1991, Lewis *et al.* 2004; but see Redpath *et al.* 2001, Tornberg & Reif 2007). By calculating food-niche breadth based on known diets (e.g., Levins 1968, Korpimäki & Marti 1995) raptors can be classified to specialists and generalists (Reif *et al.* 2001, Valkama *et al.* 2005). Specialists use a narrow range of resources and are predicted to be more efficient when foraging on their preferred food than generalist species that are able to effectively utilize outbreaks of the prey they specialize on (Korpimäki & Norrdahl 1991). However, specialists may be more vulnerable to changes in habitat quality and food resources than generalists which may be able to flexibly switch between prey species (Angelstam *et al.* 1984, Suryan *et al.* 2000). Species with overlapping niches are often thought to compete with each other (e.g., Schoener 1982), while the intensity of competition depends on the abundance of the resources. For example, raptors feeding on voles whose numbers may fluctuate fiercely in northern latitudes may not significantly compete during peak years but do so during crash years of voles (Korpimäki 1987).

Populations of the Marsh Harrier *Circus aeruginosus* have being in recovery since the 19th century throughout Europe due to improvements in the quality of their habitats, ending of illegal hunting, and bans of toxic chemicals in farming (Cardador *et al.* 2011). The recovery has also involved an expansion of its northern distribution limits. The first nesting attempt in southern Finland was in 1922 (Hildén & Kalinainen 1968), and it reached its current northernmost breeding sites at the northern end of the Gulf of Bothnia in the 1950s (Törnroos 1956).

One of the key factors in the success of the Marsh Harrier is its relatively wide diet. The Marsh Harrier is a generalist predator, being able to hunt mammals, birds, reptiles, frogs and even fish (Hildén & Kalinainen 1966, Witkowski 1989, Blanco & Hiraldo 1990, Lange & Hofman 2002). It is also known to depredate eggs of other prey species, a rare foraging habit in raptors (Opermanis 2004). Despite the fact that the Marsh Harrier is among the most successful in terms of breeding numbers among Finnish raptors, its ecology is relatively poorly known at the northern parts of its distribution, i.e., in Fennoscandia.

In the present study, we describe the diet of the Marsh Harrier in one of its northernmost breeding sites, the isle of Hailuoto in Finland, based on an analysis of prey remains. We surveyed avian prey abundances to assess whether the diet composition in the March Harrierwas related to prey abundance. Finally, we compare the focal species food niche to that of other raptors in the region to analyze (1) the potential for food competition or food-niche partitioning (Korpimäki 1987, Garcia & Arroyo 2005), and (2) the possible relationship between food-niche breadth and population trends among common raptor species in the study area.

2. Material and methods

2.1. Study area and species

Our study area, the isle of Hailuoto, is situated in the northern Bothnian Bay in the Baltic Sea (65° N, 25° E). The area of the island is about 200 km^2. Approximately 33% of the area is covered by different-aged Scots pine (*Pinus sylvestris*) forests with lichens and *Vaccinium* dwarf shrubs as an understory, 19% is covered by deciduous and 15% by mixed-wood forests. The eastern part of the island (about 10%) is covered by a peat bog and the central part is dominated by cultivated fields (about 6%) growing mainly hay, barley and oats. The long, partly overgrown lake Kirkkosalmi is situated in the middle of the island. Large reed beds *Phragmites australis*, the main breeding habitat of Marsh Harriers in Hailuoto, surround the lake that is also one of the best waterfowl habitats in the island. The sea shores are mainly open or partly open sands and dunes (about 3%) or wet meadows (2%) sporadically overgrown by reed beds (Colpaert 1998). The largest sandy shores are situated along the northern and western sides of the island, while meadows predominate in the south (Fig. 1).

Marsh Harriers return from wintering grounds to Hailuoto in late April. Nest building is initiated soon after arrival, and first eggs are usually laid

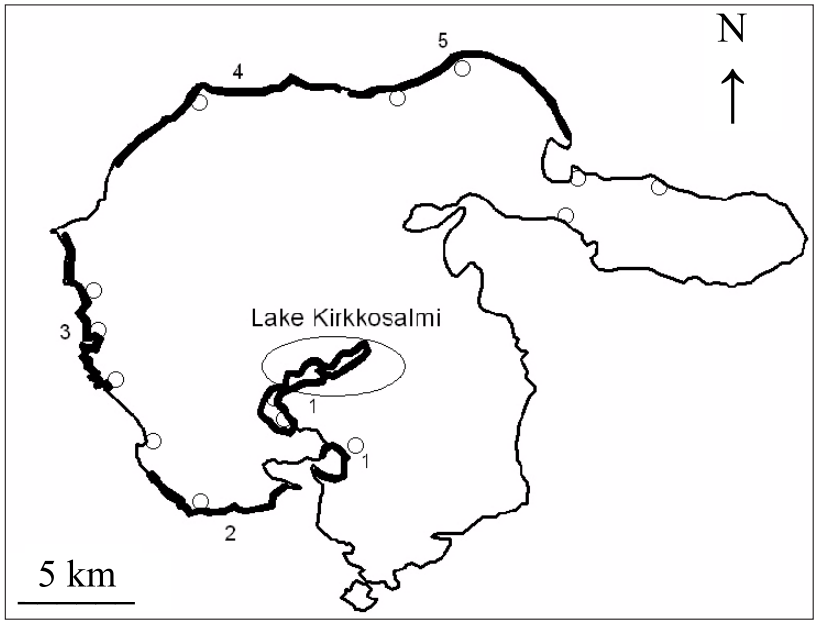


Fig. 1. Map of the isle of Hailuoto in the Bothnian Bay, Finland. In area 1 (Lake Kirkkosalmi and Viinikanlahti) shores are overgrown by reed beds. In areas 2–3, shores are variably bordered by meadows, woodland and sands with scattered reed beds, while areas 4–5 are mainly sandy shores with scattered reed beds. Thick lines represent five shorebird census areas in 2004 and 2009. Open circles show territories of the Marsh Harrier in the shore area. Oval circle shows the nesting area of Lake Kirkkosalmi.

around 11–14 May (S. Haapala unpubl. data). Chicks hatch around mid-June and first chicks fledge around 15–20 July.

Our study was conducted during 2002–2009 (excluding 2003). During these years we found 62 Marsh Harrier nests. The number of nests discovered per year varied from 3 (2002) to 22 (2009). We started nest searching around mid-May in 2002 and 2007–2009 or early June during 2004–2006. The later start in 2004–2006 probably had little effect on the number of discovered nests, as the best time for finding nests was from mid-June onwards when chicks hatched. Nests were located by observing prey delivery trips of males to the nest. Male provisioning trips were most frequent and conspicuous when chicks had hatched in mid-June (Haapala 2008). All nests were situated in reed beds near seashore, except two with a distance to the shore a few hundred meters.

2.2. Diet-composition analysis

We collected prey remains and pellets from nests three times: when a nest was found, when the chicks were ringed (approx. 10–15 July), and after the chicks had fledged. We found relatively few prey remains per nest during nestling period, while these started to accumulate during the last days of the nestling period or early fledging period when chicks still used their nests as feeding sites. Mean number of prey specimens identified was 2.8 per nest during nestling period but it was 10.1 during fledging period. This is likely to be caused by an increase in the hunting activity of the female, when she no more removes prey remains from the nest (Witkowski *et al.* 1989). We therefore divided prey samples into two groups that coarsely reflect the diet during the nestling period and that of the late nestling/early fledging period. July the 15th was a priori set as the division between the periods. Note that because the timing of breeding differed among pairs and years, this division is necessarily arbitrary. Mean wing length of chicks between 13 and 17 July was 238 5.4 SE mm, with corresponding age of 28 days (Kjijgsveld *et al.* 1998). As Marsh Harriers fledge at the age of 35 days (Krijgsveld *et al.* 1998), they apparently stayed in or nearby their nests for approx. 1–2 weeks after 15 July.

We identified 702 prey specimens in the samples. The number of prey specimens identified during the nestling and the later nestling /fledging periods were 211 and 491 individuals, respectively. We identified prey specimens using the reference collection of the Zoological Museum of the University of Oulu, Finland. Avian specimens were classified to juveniles and adults based on feather shafts whose tips were open in juveniles but closed

in adults, and color and shape. Bones of both birds and mammals were aged based on bone formation, i.e., whether the bone was porous or firm. We obtained the weight of prey specimens from Väisänen (1996).

The availability of prey animals was estimated based on shorebird and waterfowl counts carried out by the North Ostrobothnia Regional Environment Centre during May and early June in 2004 and 2009. These counts were done at protected areas of the Natura 2000 network, dispersed mainly at seashore areas (Fig. 1) by using standard count methods developed for waterfowl and shorebirds (Koskimies & Väisänen 1988). We did not collect estimates for mammals or terrestrial birds.

We used compositional analysis to compare Marsh Harrier diet to the availability of prey (Aebisher *et al.* 1993). In this method the proportion of each prey species or group in the diet and in the field is converted to log ratios by dividing the proportions of one prey type by another, and obtaining the natural logarithm of the ratio. By subtracting the log ratios of the availability of each prey pair from those of the prey used, we were able to determine whether a specific prey type was preferred over others, positive values denoting preference and negative values avoidance. By comparing log ratios of all used and available prey pairs, an anti-symmetrical matrix can be constructed. The sum of positive values on each row of the matrix gives the rank in an increasing order. We calculated the matrix for eight nesting areas, each including 1–9 nests; thus, cell values were means of these eight areas. Nesting areas denote five shorebird and waterfowl census areas, of which the areas 1, 3 and 5 were inventoried in both 2004 and 2009 (Fig. 1). The significance of values was tested by a randomization test using the Compositional Analysis Excel tool version 4.1. Due to the low number of some prey species, we grouped these prey specimens to the following categories. Firstly, grebes and ducks excluding Teal *Anas crecca* were grouped as aterfowl while Teal and Coot *Fulica atra*, as being relatively common, formed categories of their own. Other groups were waders, Common Gull *Larus canus,* Black-headed Gull *Larus ridibundus*, and terns (*Sterna paradisaea and S. hirundo* pooled). We used only waterfowl and shorebirds in the analysis because of the lack of comparable estimates from other prey types (mammals and land birds).

2.3. Diet composition of the Marsh Harrier and other raptor species

We compared the diet of Marsh Harriers with the diets of other, roughly similar-sized raptors: Hen Harrier *Circus cyaneus,* Sparrowhawk *Accipiter nisus*, Goshawk A*ccipiter gentilis*, Common Buzzard *Buteo buteo* and Peregrine Falcon *Falco peregrinus*. These species all breed on Hailuoto or nearby areas. Diet data for these other raptors were obtained from Tornberg (1997; Goshawk), Korpimäki et al. (2001; Peregrine Falcon), Reif et al. (2001; Common Buzzard) and own unpublished data (Hen Harrier and Sparrowhawk); see Appendix 1 and 2 for details.

We created a prey/raptor data matrix by first grouping prey into 21 groups: hares, red squirrel, muskrat, water vole, small rodents and shrews, weasels, waterfowl (including grebes, ducks and Coot), waders, shorebirds (gulls and terns), pigeons, the Cuckoo *Cuculus canorus*, raptors, owls, woodpeckers, corvidae, thrushes, small passerines, reptiles, frogs, fish, and insects. We then divided prey samples of each raptor species into these categories. For comparative reasons we used only samples collected from nests after fledging.

We analyzed the diet similarity by using multidimensional scaling (MDS) with proxscal procedure available in the SPSS 14.1 software. MDS applies Euclidean distances from standardized values of the matrix, and produces a two-dimensional space according to two main dimensions of data variation. The distance between the points is comparable to the similarity of the diets between raptor species.

2.4. Food-niche breadth and population trends

We categorized the ten raptor species as being either ommon or are based on their population numbers in the 1980s (Saurola 2008). We aimed to distinguish between species that (i) were not known to have remarkably declined or were not of conservation concern and (ii) were rare compared to their historical density decades earlier (Väisänen *et al.* 1998). For the latter, causal relationships between

diet and population growth might be better perceivable. Because the recent rapid population growth of some of these species, they may no longer be considered rare. With common we refer to species whose density in the beginning of raptor monitoring was in some thousands of pairs.

We analyzed the species-specific relationship between food-niche breadth and population trends using data collected in or near the study area during 1982–2006; for five species, see above. We collected additional data for the Honey Buzzard *Pernis apivorus* (Mikkola & Itämies 1972), Kestrel *Falco tinnunculus* (Korpimäki 1985), Golden Eagle *Aquila chrysaetos* and the White-tailed Sea Eagle *Haliaeetus albicilla* (Sulkava *et al.* 1997, 1998), complemented with our own unpublished data.

To calculate the diet width we applied Levins index (Levins 1968)

$$B = \Sigma\ 1/p_i^2 \qquad (1)$$

where p_i is the proportion of the *i*th prey or prey group. Population trends for raptor species during 1982–2006 were obtained from Saurola (2008). We applied a general linear model to examine the relationship between diet width and population change using R 2.13.1 (R Development Core Team 2012) software.

3. Results

3.1. Breeding-season diet of the Marsh Harrier

The Marsh Harrier diet during the nestling phase from hatching until late nestling was composed mainly of birds but in terms of biomass their share was less important. Later in the breeding season the proportion of birds declined and the proportion of mammals increased accordingly (Table 1). Passerine birds dominated the diet during the nestling phase, yet their biomass accounted only for 13.5% of the diet. Waterfowl (ducks and the Coot) and shorebirds (waders, gulls and terns) made up almost 50% of the biomass, ducks being the most important prey group.

During the fledging phase the proportions of the prey groups evened out. Waterfowl and shorebirds became the most important group. The most important prey species among the waterfowl were the Coot and the Teal (Table 1). Shorebirds (waders, gulls and terns), and especially their chicks, made up a significant part of the diet. Chicks constituted 82% of all gulls and terns found in the remains during the fledgling period. Generally, about 60% of bird prey was pulli or fledglings during both periods. All grouse, except one adult Willow Grouse *Lagopus lagopus,* were presumably chicks of the Black Grouse *Tetrao tetrix,* which is a stable part of the Marsh Harrier diet on Hailuoto. During the fledging period, passerine birds were a rather insignificant group in terms of weight, while the share of mammals, primarily muskrats *Ondatra zibethica*, increased both by number and weight from nestling to fledging periods. Hence, muskrats that were all juveniles were the most important single prey species in late summer. Leverets of *Lepus* species (*L. timidus* or *L. europeaeus*) were also important, accounting for 10–12% of the diet by weight during both periods. Small mammals, reptiles, amphibians and fish were rare in the diet. The mean prey weight increased two-fold from the nestling to the fledging period. The

Table 1. Proportions (%) of different components of the diet of the Marsh Harrier in the breeding seasons 2002–2009 in Hailuoto, northern Finland.

	Nestling period		Fledging period	
Group	Number	Weight	Number	Weight
Mammals, total	6.6	16.9	25.8	38.1
– Muskrat	1.4	5.0	14.9	24.3
– Leverets	2.8	9.9	7.5	12.2
– Other mammals	2.4	2.0	3.3	1.5
Birds, total	92.4	80.7	73.0	60.7
– Grebes	–	–	0.6	1.8
– Herons	0.5	4.1	–	–
– Ducks	10.9	33.8	8.7	10.9
– Grouse (juv.)	2.8	7.0	7.9	9.2
– Coot	0.9	3.4	8.5	14.4
– Waders	8.5	11.0	5.7	3.2
– Gulls and terns	3.3	4.6	17.1	16.1
– Passerines	63.5	13.5	20.9	2.5
– Other birds	1.9	3.3	3.5	2.7
Reptiles	0.5	1.7	0.2	0.0
Fish	0.5	0.7	1.0	1.1
Total (*n*)	211	–	491	–
Mean weight	–	143	–	307

Table 2. Cross-tabulation of water- and shorebirds determined by a compositional analysis. The first row for each species or group denotes the size and direction of the preference, and the lower row denotes SE. The number of positive values in the upper row indicates the preference rank of prey species or group, with peak values denoting the most preferred species.

	Teal	Other. waterf.	Coot	Waders	Black-h. Gull	Common Gull	Terns	Rank
Teal		–1.212	–2.124	–1.987	–0.101	0.379	–0.731	1
		1.304	1.680	1.264	2.222	1.466	1.081	
Other waterf.	1.212		–0.912	–0.775	0.768	1.591	0.480	4
	1.304		1.105	0.432	2.322	1.408	1.233	
Coot	2.124	0.912		0.138	1.984	2.503	1.393	6
	1.680	1.105		1.277	3.360	1.216	1.933	
Waders	1.987	0.775	–0.138		1.266	2.366	1.255	5
	1.264	0.432	1.277		2.269	1.590	1.139	
Black-h. Gull	0.101	–0.768	–1.984	–1.266		0.381	0.006	3
	2.222	2.322	3.360	2.269		2.958	1.564	
Common Gull	–0.379	–1.591	–2.503	–2.366	–0.381		–1.110	0
	1.466	1.408	1.216	1.590	2.958		1.591	
Terns	0.731	–0.480	–1.393	1.255	0.006	–1.110		2
	1.081	1.233	1.933	1.139	1.564	1.591		

compositional analysis did not detect significant differences between consecutively-ranked species indicated by the randomization test (all $p > 0.05$) between consumption and availability of shorebirds and waterfowl (Table 2).

3.2. Diet of the Marsh Harrier compared to other raptor species

The diet of the Marsh Harrier was most similar with that of the Sparrowhawk, the distance between the arithmetic means being 0.545 (Fig. 2). Distances to the Peregrine Falcon (0.710), Hen Harrier (0.756) and Common Buzzard (0.832) were of similar magnitudes, while it was considerably longer to the Goshawk (1.702). In one year the diet of the Common Buzzard was relatively close to that of the Marsh Harrier. This year was a crash year for voles, forcing Common Buzzards to shift from small mammals to thrushes and small passerines. Correspondingly, the Hen Harrier diet resembled that of the Common Buzzard during the two years when Hen Harriers specialized on *Microtus* voles.

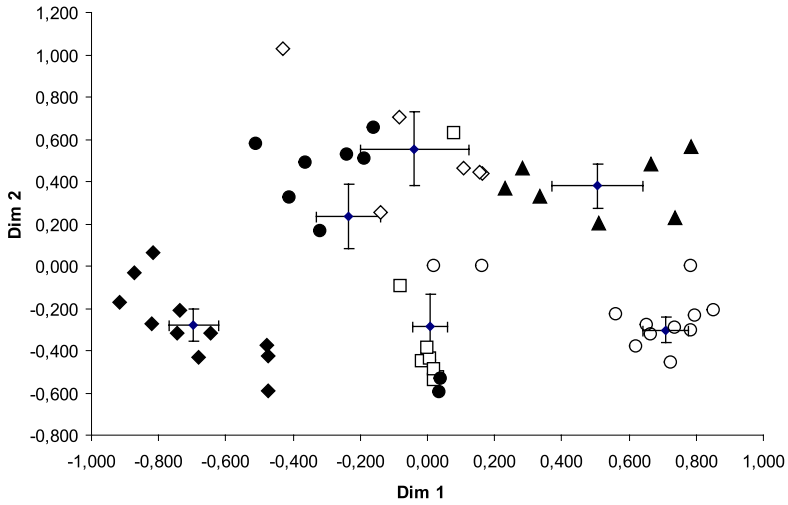


Fig. 2. Multidimensional scaling for six raptor species: Marsh Harrier (filled triangles), Hen Harrier (filled circles), Sparrowhawk (open diamonds), Goshawk (filled diamonds), Common Buzzard (open squares) and Peregrine falcon (open circles). Each symbol represents an annual food sample of a specific species. Black dots with error bars (95%CI) denote the arithmetic means of the locations of each species diet in the two-dimensional space.

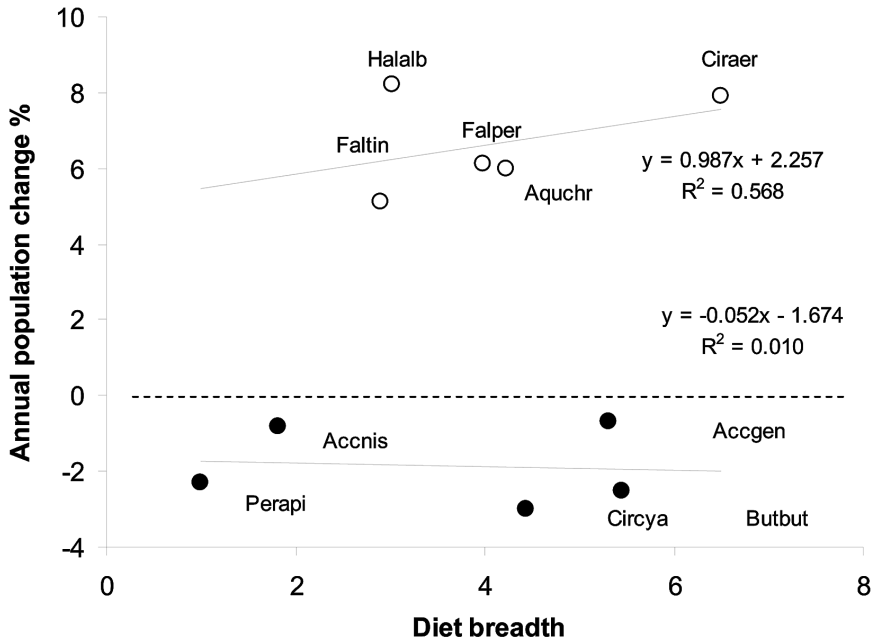


Fig. 3. Relationships between diet breadth and annual population change for ten raptor species (Ciraer = Marsh Harrier, Circya = Hen Harrier, Accnis = Sparrowhawk, Accgen = Goshawk, Butbut = Common Buzzard, Perapi = Honey Buzzard, Aquchr = Golden Eagle, Halalb = White-tailed Eagle, Faltin = Kestrel and Falper = Peregrine Falcon). The first five species are common (filled circles) and the remaining five are rare (hollow circles), the classification being based on their abundance according to raptor monitoring in the 1970s.

3.3. Food-niche breadth and population change in raptors

We plotted diet breadth against the annual mean of the population change for common and rare species separately. Common species showed weak negative population development for the diet breadth, while it was slightly positive in rare species. However, neither of these trends were significant. Instead, there was a significant difference between these two groups in that rare species had higher population growth than common ones (GLM; $F_{2,10} = 56.48$, $p < 0.001$; Fig. 3, Table 3).

4. Discussion

The diet of the Marsh Harrier was diverse, consisting of prey species from both aquatic and terrestrial habitats. Passerines dominated the diet during the early nestling period by number, whereas ducks were the most important prey group by weight. Later in the breeding season the importance of mammals, grouse, the Coot and gulls increased. When compared to other diet studies, our results deviate mostly in terms of the share of small mammals. While these account for between 20 and 70 percent in other studies (Hildén & Kalinainen 1966, Witkowski 1989, Lange & Hofman 2002), we found them to account for a low percentage. Clearly, young muskrats had taken the role of small mammals in our study area. We did not find any small mammals in the pellets that tend

Table 3. Parameter estimates of a general linear model explaining population changes in ten raptor species. Species classification to rare and common species was used as a fixed variable and diet breadth as a covariate. For details, see text.

Parameter	Estimate	SE	*t*	*p*
Intercept	–10.656	1.486	–7.173	<0.001
Species	8.473	0.813	10.422	<0.001
Diet breadth	0.090	0.252	0.357	0.731

to give them higher percentages than prey remains (Lange & Hoffman 2002). The reason for that might be an exceptionally long-lasting low population phase in the field vole *Microtus agrestis* and the water vole *Arvicola terrestris* during the study years in Hailuoto.

Passerines commonly dominated the avian part of the diet in Marsh Harriers, accounting for about 20–30% in the diet (Hildén & Kalinainen 1966, Witkowski 1989, Lange & Hofman 2002). In our sample from the nestling phase, passerines accounted for over 60% of prey items by number, which may be due to a low share of small mammals in the diet that are sometimes replaced by passerines in the Marsh Harrier diet (Lange & Hoffman 2002) but also due to the timing of our diet sampling. If prey are sampled only after fledging, the diet composition differs from that obtained in the middle of the nestling period when only the male delivers prey to the nest. Witkowski (1989) observed that males hunted mainly in terrestrial, whereas females predominantly hunted in aquatic habitats. During the breeding season, males seem to have larger home ranges than females, enabling them to encounter terrestrial species more often, while females hunt near their nests in wetlands, which could be one reason for the habitat separation and hence for different prey choice between the mates (Cardador *et al.* 2009). Another explanation may be a change in the availability of different prey species. Young growing individuals of muskrat, leveret, grouse and gull become more beneficial and visible prey for hunting raptors in mid and late summer, while at the same time fledglings of passerines disperse and are more difficult to detect. Increased food demand by harrier chicks may also force parents to hunt larger prey, especially if small mammals are not abundant (Cardador *et al.* 2012), which was the case in Hailuoto.

The compositional analysis showed that Marsh Harriers hunt opportunistically among waterfowl and shorebirds. We found no significant differences between successively-ranked prey types, except for the first (the Coot) and last rank (the Common Gull). Coots were mainly found in nests situated in the luxuriant Lake Kirkkosalmi where we also found the highest density of nesting Marsh Harriers. Also many earlier studies have reported the Coot to be the most important waterfowl species in the diet of the March Harrier (Hildén & Kalinainen 1966, Witkowski 1989, Blanco & Hiraldo 1990, Lange & Hofman 2002). The low rank of gulls may reflect the aggressive nest defense in gull colonies against Marsh Harrier attacks, although some pairs subsisted on juvenile gulls living on sandy dunes of the shores of Hailuoto.

We did not find diet overlap between the Marsh Harrier and other raptors. This suggests that the Marsh Harrier has adopted a relatively distinct food niche as compared with other raptors. Although the Sparrowhawk had quite similar diet to the Marsh Harrier, they may not significantly compete with each other as they generally occur in different habitats (forest vs. open). The same is true for the Common Buzzard that shares the diet of Marsh Harriers in years of low vole abundance (Reif *et al.* 2001). During low vole years Marsh Harriers and Buzzards may potentially compete for the same food resources. However, this was not apparent in Hailuoto, as the Common Buzzard is an extremely rare breeder in the island. Competition may also be avoided through habitat segregation (Garcia & Arroyo 2005). Although the breeding and hunting habitats of the Marsh Harrier coincide virtually only with those of the Hen Harrier, these two species have distinctive food-niches. The fairly high proportion of grouse chicks and leverets in the Marsh Harrier diet, on the other hand, suggests that in our study area the species frequently hunts in forested habitats, thereby potentially overlapping with the Goshawk and the Sparrowhawk. In central and southern Europe, however, Marsh Harriers almost exclusively use open habitats (Witkowski *et al.* 1989, Lange & Hoffman 2002, Cardador *et al.* 2011). Interestingly, although being a common breeder in Hailuoto, the Goshawk has not been found to kill any Marsh Harriers, young or adult (authors pers. obs.).

We found no correlation between population trends and food-niche -breadths of either rare or common raptor species, but the former had significantly positive and steeper trends than common species, which all had negative trends. Among the ten species, the Marsh Harrier had the widest diet breadth and the second-fastest population growth. The Marsh Harrier is also quite opportunistic in its foraging habits and may avoid interspecific competition either for food or breeding sites. The latter may be an important limiting factor especially

among forest-living raptors (Hakkarainen *et al.* 2004). The Marsh Harrier has benefited from the expansion of reed beds, their preferred breeding habitat (Stanevicius 2004, Haapala 2008), which has taken place on the shores of Hailuoto due to enrichment of waters and cessation of cattle grazing (since 1956 in Hailuoto). One reason for the success of the Marsh Harrier may be the generally high productivity of their breeding habitats where polygyny may be an effective breeding strategy (Altenburg *et al.* 1982). We found apparent cases of polygyny mostly on Lake Kirkkosalmi, where the polygyny rate varied among years between 0% and 10%.

The strongly-increasing tare species have benefited from vast areas having become vacant during the largely human-caused population collapses in the 1960s and 1970s (Newton 1979, Stjernberg 2003). The Marsh Harrier, on the contrary, is a newcomer that has adopted an open niche, the shores. In spite of its wide diet breadth, it is also well adapted to man-made habitats and may even benefit from certain human activities (Witkovski *et al.* 1989, Lange & Hoffman 2001, Cardador *et al.* 2011). The same concerns some other species such as the Peregrine Falcon although its natural breeding habitats, open bogs, are declining due to draining and turf industry. Among common species that showed negative populations trends, the Goshawk probably suffers from loss of habitat (mature forests) and main prey (forest grouse; Widén 1997). The Honey Buzzard may face the same problem as it seems to be even more engaged with mature forests than the Goshawk (P. Byholm, pers. comm.). Vole specialists the Common Buzzard and Hen Harrier on the other hand, may have been adversely affected by the irregularity of vole fluctuations. Especially if spring densities of voles remain low, e.g., due to mild winters with poor snow cover not providing sufficient shelter for voles, the reproduction of vole specialists may fail (Solonen & Karhunen 2002, Sundell *et al.* 2004, Lehikoinen *et al.* 2009). It is noteworthy that the Marsh Harrier, although basically having a preference for voles (e.g., Lange & Hoffman 2002), seems not to depend on voles either in spring or in summer, thus avoiding many dangers the vole specialists may face. The Marsh Harrier has also other sources of food, such as eggs of other birds that are plentiful in late spring (Opermanis 2001) and, as our results showed, a wide variety of birds.

The observed diet breadth of the Marsh Harrier suggests flexibility against environmental changes, although this did not seem to explain the success of the examined ten raptor species. Some of the successful, rare species have apparently adapted to man-made environments, even urban areas, while many losers live in mature forests. Northern forests, although still covering large areas, may still become poorer for top predators although human-modified landscapes may provide an increasing food source for them (Cardador *et al.* 2011).

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Hailuodossa pesivien ruskosuohaukkojen ravinto suhteessa muihin Pohjois-Suomessa pesiviin päiväpetolintuihin

Ruskosuohaukka on yleistynyt voimakkaasti Pohjois-Suomessa viimeisten vuosikymmenten aikana. Tutkimme ruskosuohaukkojen ravinnonkäyttöä ja -valintaa Hailuodossa vuosina 2002–2009. Lintujen osuus vaihteli pesimäkauden kuluessa välillä 73–93 %. Pienten varpuslintujen osuus oli pesäpoikasajalla 63 %, mutta saaliin kokonaismassasta niiden osuus oli vain 13 %. Nisäkkäistä piisami oli tärkein, sen lukumääräosuuden ollessa 15 % ja massaosuuden ollessa 24 %.

Rakenneanalyysin mukaan ruskosuohaukat saalistivat vesi- ja rantalintuja opportunistisesti. Moniulotteisen skaalauksen perusteella ruskosuohaukan ravintolokero oli erilainen kuin viiden muun petolintulajin lokerot. Ravintolokeron leveys ja vuotuinen populaatiokasvu kymmenellä päiväpetolintulajilla osoitti, että ruskosuohaukan ravintolokero oli levein ja populaatiokasvu toiseksi nopein. Vuotuinen populaation muutosnopeus oli positiivinen lajeilla, jotka olivat vähälukuisia petolintujen kannanseurannan alettua 1970-luvulla, mutta negatiivinen tuolloin runsaslukuisilla lajeilla.

Merkitsevää suhdetta ei ollut havaittavissa ravintolokeron leveyden ja populaation vuotuisen muutoksen välillä vähä- tai runsaslukuisilla lajeilla.

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Appendix 1. Prey species identified in food samples collected from the Marsh Harriers' nests in Hailuoto, northern Finland during 2002–2009. Numbers are individuals.

Taxon		2002	2003	2004	2005	2006	2008	2009	Total
Erinaceus europaeus juv.	500		1						1
Sciurus vulgaris	270					1			1
Arvicola terrestris	150	3	1	1	1	2	1	3	12
Ondatra zibethica juv.	500		9	5	10	41	9	2	76
Microtus agrestis	20		1						1
Microtus/*Myodes*	20				1		2	2	5
Lepus sp juv.	500	3	6	4	9	10	6	5	43
Mustela erminea	156						1		1
Large mammal sp.	500			1					
Mammals, total		**6**	**18**	**11**	**21**	**54**	**19**	**12**	**141**
Podiceps cristatus	930	1							1
Podiceps sp.	900		1			1			2
Botaurus stellaris	1230							1	1
Anser anser juv	500						1		1
Anas platyrhynchos	1,100						2	1	3
Anas crecca	300		4	1	3	12	9	3	32
Anas querquedula	300		1		1	1	1		4
Aythya marila	603							1	1
Aythya fuligula	720		1	1	1	1	1	2	7
Bucephala glangula	750			1			1		2
Anseriformes	300		2	2	3	5	4		16
Accipiter nisus	200					1		1	2
Falconiformes	200		1						1
Lagopus lagopus	585						1		1
Tetraonidae juv.	350	3	5	1	16	4	13	2	44
Fulica atra	520			2	21	10	6	5	44
Haematopus ostralegus	480			2		1	1		4
Charadrius hiaticula	58					1			1
Vanellus vanellus	206		1	1	4				6
Tringa totanus	107	1	2		3	1			7
Tringa nebularia	180			1	2	1	1		5
Tringa glareola	60						1	3	4
Tringa ochropus	80							1	1
Scolopax rusticola	300	1			2			1	4
Gallinago gallinago	95				1				1
Philomachus pugnax	150	1			1				2
Charadriformes	206		2	2	1	2	2	2	11
Larus ridibundus	265	1	9	5	6	4	5	1	31
Larus canus	415	1	8	1		3	23	0	36
Sterna sp.	110		13	3	1	2	4	1	24
Columba palumbus	500		1					1	2
Columba sp.	300							1	1
Asio flammeus	315						1		1
Aegolius funereus	123					1			1
Dendrocopos major	88		1		1	1	2		5
Hirundo rustica	19						1		1
Anthus trivialis	23			2	17	1	6	14	40
Anthus pratensis	18	1			1				2
Motacilla alba	20					1			1
Motacilla flava	20				1				1
Bombycilla garrulus	56				2				2
Saxicola rubetra	16				1		1		2
Turdus merula	101				1		1		2

Appendix 1, continued

Turdus pilaris	105			2	2				4
Turdus iliacus	60	1		1	6		1		9
Turdus philomelos	69			2	5				7
Turdus sp.	80		3	2	2	1	1		9
Acroc. schoenobaenus	12			6	2				8
Sylvia borin	20		2		5				7
Phylloscopus trochilus	9	1		1	11	3	2	1	19
Ficedula hypoleuca	13		1		2				3
Muscicapa striata	16	1			2				3
Parus montanus	11							1	1
Parus caerulaeus	11			1	3		1		5
Parus major	20		1	1	8		2	2	14
Lanius collurio	29							1	1
Pica pica	235				3			1	4
Corvus corone cornix	525					1			1
Fringilla coelebs	22	3	2	1	18	2	6	2	34
Fringilla montofringilla	22				2				2
Carduelis spinus	13	1	1		3	1	1	4	11
Carduelis chloris	29	1			1		1		3
Pyrrhula pyrrhula	31				3			1	4
Carpodacus erythrinus	23				1				1
Loxia curvirostra	41	1		1				1	3
Emberiza citrinella	31						1		1
Emberiza schoeniclus	19				4			1	5
Passeriformes	20	1	6	2	9	3	2	4	27
Middle-sized bird	210		1	2		2	1		6
Large bird	510				1	1			2
Birds, total		**20**	**69**	**47**	**183**	**68**	**107**	**60**	**554**
Lacerta vivipara	9	1							1
Vipera berus	500							1	1
Esox lucius	500				2				2
Abramis brama	300						1		1
Unidentified fish	200		1			1	1		3
Total		**27**	**88**	**58**	**206**	**123**	**128**	**73**	**703**

Appendix 2. (a) Sample size, number of study years and number of nests collected for six raptor species in Hailuoto and nearby areas. (b) Numbers of water and shorebirds found in the diet of Marsh harriers and counted in five different areas in Hailuoto during 2004–2009, denoted by numbers 1–5 (see Fig. 1). Diet data of areas 1, 3 and 5 were split into two periods (2004–06 and 2007–09) according to counts done in 2004 and 2009.

(a) Variable	Sparrowhawk	Common Buzzard	Marsh Harrier	Hen Harrier	Goshawk	Peregrine Falcon
No prey	240	1,869	706	197	1,224	805
Nests	9	113	69	9	85	41
Years	6	8	7	7	11	9

(b) Area	Teal	Other wf	Coot	Waders	Black-h. Gull	Common Gull	Terns
Birds in diet							
1		25	17	36	23	9	3
2	0	1	0	7	4	1	2
3	2	5	0	7	4	1	2
4	0	3	0	1	1	24	3
5	0	8	1	5	0	2	1
Total	**27**	**34**	**37**	**43**	**18**	**31**	**13**
Birds counted							
1	73	373	61	155	140	41	67
2	7	194	0	92	0	31	82
3	40	291	5	156	141	210	154
4	28	156	0	122	6	146	191
5	23	221	0	125	396	46	305
Total	**171**	**1,235**	**66**	**650**	**683**	**474**	**799**

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Environment Agency



Cooling Water Options for the New Generation of Nuclear Power Stations in the UK

SC070015/SR3

Better regulation programme

Evidence Directorate

Executive summary

The consideration of new nuclear power stations is split into two phases. The first phase addresses generic design matters (namely, acceptability of candidate nuclear power station designs) and the second deals with site-specific applications for permits under the Environmental Permitting (England and Wales) Regulations 2010.

The Environment Agency, Health and Safety Executive's Nuclear Installations Inspectorate and the Office for Civil Nuclear Security are currently assessing candidate designs of new nuclear power stations. The Environment Agency is exploring the environmental effects of candidate designs based on a generic site description. A statement about the acceptability of that design for a generic site in England and Wales will be provided.

Cooling water is required to remove “waste heat” from power stations regardless of whether the stations are nuclear or conventional. A nuclear power station has a typical thermal efficiency of 25-33% (compared to around 40% for a modern coal-fired station) and hence a 1,000 megawatt electric (MWe) nuclear station would typically generate up to 2,000 megawatts of low-grade waste heat. The reasons for this apparent wastage are explained. The report also explores cooling water options for new reactors and evaluates their potential environmental impacts in terms of effects on biota, and thermal, chemical and radionuclide pollution. The findings are focused on, but not confined to, nuclear plants and will have general applicability to other large (above 1,000 MWe) thermal power station projects.

This report is based on publicly available information and publications and the experience of the authors. It provides an overview of power station cooling water systems in use in the UK and abroad. Details of cooling water options for new nuclear power stations in the UK are given. Cooling water system design (direct and indirect cooling water systems, intake and outfall designs), how the design affects the performance of the cooling option and issues such as temperature differentials between water intake and discharge are discussed.

An overview of environmental issues associated with cooling water systems of nuclear and other large power stations is presented. Issues arising from water abstraction and discharges are discussed. These include fish and invertebrate intake and impingement on filter screens, effects of passage of planktonic and small life-stages through the cooling system, thermal, chemical, radionuclide pollution, and effects of cooling tower emissions to air. The report also discusses different mitigation measures (such as intake location, intake screen designs to minimise impacts of entrapment, entrainment and impingement, plume abatement techniques to minimise effects of plume formation). Implications of combining conventional liquid discharges within the cooling water discharge are also considered.

Environmental issues with specific cooling water options are identified and explained. The report evaluates cooling water options in terms of environmental concerns (including water demand and energy efficiency) and assesses the best options for use in different types of water environments (coastal, estuarine and fresh waters). Effects of climate change on the choice of cooling water options are also briefly considered.

It is likely that new UK nuclear stations will be built on coasts or estuaries. A key question at the outset of the study was whether direct cooling (also known as “once-through”) can still be considered Best Available Technology (BAT) for large coastal and estuarine power stations, as set out in the European Commission’s BAT reference

document on industrial cooling systems (BREF-Cooling, 2001). While this has recently been challenged in relation to a proposed 2,000 MWe combined cycle gas turbine (CCGT) power station at Pembroke, the findings of our study indicate that direct cooling can be BAT for estuarine and coastal sites, provided that best practice in planning, design, mitigation and compensation are followed. The potential BAT-status of direct cooling has essentially been preserved owing to improved understanding of survivability of the entrainment process, and substantial developments in impingement mitigation techniques since the BREF was written. As per the BREF advice, there may remain cases where, even with the application of best practice, residual impacts would be unacceptable. In these cases, seawater cooling towers would be used. BREF advocates the use of dry-cooling methods only where water is in extreme short supply; this advice remains appropriate. These conclusions are generic and site specific applications will be assessed individually.

The findings are applicable to both nuclear and conventional power stations. We conclude that direct cooling may be the best option for some nuclear power stations. A summary of impacts from the various cooling is summarised below:

Environmental concern	**Direct cooling**	**Cooling towers**		
		Natural draught (wet)	**Mechanical draught (wet)**	**Natural draught (dry)***
Generation efficiency	High efficiency Uses less fuel so lower aerial emissions	Typically 0.5 - 1.5% less efficient than direct cooling	Typically ~2% less efficient than direct cooling	Lowest efficiency 2 - 3% less efficient than direct cooling
Complexity	Low	Moderate	High	Very high
Water abstraction	High	Moderate/low	Moderate/low	None
Abstraction effects	Site-specific -depends on characteristics of receiving waters			
Water consumption	None on-site	Moderate	Moderate	None
Visible plumes	None	Moderate	Moderate/low	None
Ground fog & icing	No icing. Local fog plume over shoreline discharges	None	Possible	None
Visual impact	Occasional foam or 'slick' at outfall	High	Moderate	High
Noise	None	Low	Moderate	Low/none
Discharge effects	Site-specific -depends on characteristics of receiving waters			
Waste disposal to landfill**	None if using fish recovery & return***	Moderate	Moderate	Moderate/none
Land use on-site****	None/low	Moderate/high	Moderate	High

* See sections 3.1.9 and 3.1.11

** Wastes from wet towers are mainly silt (non-hazardous); from dry towers, glycol (non-hazardous), if used

*** See section 7.2.3 'Consenting Issues' and section 6.1.6 'Biota recovery and return techniques'

**** This covers buildings and structures only and does not include spray ponds or cooling canals

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1 Introduction

1.1 Purpose of study

The Health and Safety Executive (HSE) and Environment Agency[1] are working together to make sure that any nuclear power station built in the UK meets high standards of safety, security, environmental protection and waste management (Generic Design Assessment, GDA). There is a need to investigate cooling water options for new reactors and evaluate the environmental impact of these in terms of thermal, chemical and radionuclide pollution, and impact on biota. This document draws together information that will assist the regulatory agencies in this process.

1.2 Background

Regulation of the nuclear power industry in the United Kingdom is the joint responsibility of the Environment Agency and the Scottish Environmental Protection Agency (SEPA) (environmental) and the Health and Safety Executive (nuclear health and safety), that now includes the Nuclear Installations Inspectorate. The Government is committed to allowing the construction of new nuclear power stations provided they are subject to the normal planning process for major projects (under a new national planning statement) and provided also that they receive no public subsidy. The Government will complete the drafting of a national planning statement and put it before Parliament and, if approved, clear the way for planning applications for new nuclear power stations. This policy creates new challenges for the regulatory agencies given the variety of options available, since the development of nuclear power stations is now open to commercial competition. Hitherto all nuclear design and construction has been under effective government control, via the former Central Electricity Generating Board (CEGB) or, until 1996, the state-owned Nuclear Electric (NE) plc. Sizewell B was commissioned before NE was sold-off as British Energy.

A Strategic Siting Assessment of potential sites for nuclear new build has been carried out using exclusionary and discretionary criteria which were consulted upon publicly. This included a criterion on access to suitable sources of cooling. The Environment Agency advised the Government on whether sites were potentially suitable against this criterion. The public has been consulted upon whether sites are potentially suitable and the Government is currently considering responses to the consultation. Sites which are potentially suitable for deployment by 2025 will be listed in a Nuclear National Policy Statement. Should individual applications come forward for development consent, the Environment Agency will consider cooling as part of licensing with site specific detail.

Most people would be astonished at the demand for cooling water (CW) imposed by a nuclear plant, or indeed any other thermal power station. Laws of thermodynamics dictate that for every megawatt of power generated, up to two megawatts must be discarded as low-grade (low temperature) waste heat. In fossil-fuelled plants such as combined-cycle gas turbine (CCGT) stations and the new generation of supercritical coal plants significantly higher efficiencies (above 45 per cent) can be achieved, and in some cases low-grade heat can be put to use in district heating schemes, commercial horticulture or aquaculture. For nuclear stations, however, thermal efficiencies remain relatively low (around one-third efficient) and opportunities for constructive use of waste

[1] See Joint Regulators website: http://www.hse.gov.uk/newreactors (viewed 12/04/09)

heat around isolated sites are fewer. Thus, a nominal 1,000 MWe nuclear generating station must discard nearly 2,000 MW into the environment as waste heat.
Heat disposal from thermal power stations can be into the atmosphere via cooling towers, via once-through direct cooling systems into surface water bodies and thence into the atmosphere or via hybrids that combine both methods. Direct cooling is the most efficient in terms of energy use, and therefore in terms also of greenhouse gas emissions. All UK nuclear stations built before today, and a large number of fossil-fuelled stations, discard their heat to water via direct cooling, the key reason favouring construction on the coast or estuaries.

Under the EU Integrated Pollution Prevention and Control 'IPPC' Directive (96/61/EC), the Environment Agency is obliged to consider, for designated installations, whether the technologies and techniques used by the developer would be Best Available Techniques or BAT. Technical guidance on BAT is given in BAT Reference or BREF notes. The European BREF document on industrial cooling systems (BREF Cooling; adopted December 2001) considers water cooling as the preferred option (as it helps reduce emissions of greenhouse gasses) and defines direct cooling as a BAT for large combustion plants in coastal locations, provided that the aquatic ecosystem is not adversely impacted. While nuclear power stations are not within the scope of the IPPC Directive, the BREF notes are useful to inform the Environment Agency's discharge consenting process. OSPAR (Oslo and Paris Conventions) also requires the use of BAT when considering prevention and elimination of marine pollution. OSPAR covers discharges of radioactive substances[2].

More recently, however, the validity of the BAT definition of direct cooling has come under challenge, owing to what some believe may be unacceptable environmental impacts on the source and receiving water bodies. Pressure on this issue has emanated largely from activities in the USA, where impacts arising from abstraction of cooling water are regulated under the Clean Water Act s.316(b). Legal actions brought against the US Environmental Protection Agency (EPA) by the pressure group Riverkeeper, Inc. have forced the EPA to act on this issue. In its 2004 decision in Riverkeeper v. EPA (Riverkeeper I), the Second Circuit upheld the EPA's Phase I regulations, which set closed-cycle cooling systems (wet or dry tower-cooling) as BTA (best technology available) for new power plant cooling facilities[3]. In this case, the finding relates to effects of the abstraction only, and not of the discharge of heated effluents into the environment. In the UK, the validity of direct cooling as BAT for a new 2,000 MWe power station proposal falling within a Natura 2000[4] site, was investigated on behalf of the Countryside Council for Wales (Cambrensis, 2008). A key conclusion from the Cambrensis report was that, since the latest information to the 2001 BREF Cooling note was published in 1997, it is now out of date given developing techniques in indirect cooling. However, the same could be said of environmental mitigation techniques against abstraction impacts, an aspect which is not considered in the Cambrensis report. To give proper consideration to this issue, a thorough understanding of the technical background and the types and levels of environmental impacts are required, along with an appreciation of recent developments in cooling technology and associated mitigation techniques.

Effects upon the aquatic environment from direct cooling water systems, that is those that do not use cooling towers, relate primarily to two causes:

- The incidental capture of organisms contained in the abstracted cooling water stream: these may be divided into two components, the first caused

[2] http://www.ospar.org/content/content.asp?menu=00220306000063_000000_000000
[3] *Riverkeeper I*, 358 F.3d 174, 194 (2d Cir. 2004).
[4] Milford Haven Special Area of Conservation

by impingement of fish and invertebrates on cooling water filter screens and the second, entrainment of mainly planktonic stages of fish, invertebrates and microscopic plants which pass through the screens through the cooling circuit, before being discharged back to the wild via the thermal discharge.

- The effects of the thermal discharge, which sometimes contains residual oxidants and byproducts from use of biocides (mainly chlorine-based) in the receiving water bodies.

Where cooling towers are used, these impacts are diminished but arise, for example, from cooling tower plume emissions and visual impact and increased energy use (or decreased efficiency of the power plant). This report helps to explain why and how water is used for power station cooling, what the technical options are and the main design considerations, types and levels of environmental impact and mitigation techniques now available.

1.3 Terms of reference

The following objectives were established for this study:

1. to give a brief overview of all power station cooling water systems in use in the UK and abroad;
2. to identify and give details of all cooling water options for new nuclear power stations in the UK;
3. to give an overview of the generic environmental issues associated with cooling water systems of nuclear power stations (for example thermal, chemical and radionuclide pollution, fish and invertebrate intake);
4. to identify and explain any environmental issues associated with specific cooling water options;
5 to evaluate the cooling water options in terms of environmental concerns and assess the best options for different types of water body (coastal, estuarine and freshwater).

1.4 Sources of Information

The study uses only publicly available documents and references, all of which are listed in the text or in the reference list at the end of the report. Where internet sources have been used, the dates on which these were accessed are given. A number of internal CEGB and power industry reports have been cited, which should be available from the British Library but are also held by Jacobs Engineering, Southampton Office.

1982), equivalent to 2.7-4.6 tonnes per year. On this basis, loss due to impingement at the power station equates to a percentage or multiples of the annual diet of a grey seal.

The dietary consumption of piscivorous seabirds has been estimated by Camphuyson *et al. (*in Turnpenny, 2002) who investigated the number of seabirds that would be supported by fish discarded from fishing vessels. Such discards comprise undersized fish and those of non-target species that are thrown back to sea. Their figures provide a basis for assessing the number of seabirds supported by fish removed by a power station. They are based on a hypothetical 1,000 g seabird, with a daily energy intake (3x basal metabolic rate) of around three times 600 kJ per day, or 6.57×10^5 kJ y^{-1}. The calorific value of fish was estimated at 5 kJ g^{-1} for roundfish, 4 kJ g^{-1} for flatfish. Using these figures, fish removed by the power station can be equated to the "lost food" of a number of 1,000 g seabird equivalents. Taking the average value of 4.5 kJ g^{-1}, the 0.4 tonnes of fish removed by Fawley Power Station was estimated to equate to 1.8 x 106 kJ y^{-1}, or the dietary requirements of nearly three 1,000 g seabirds (Turnpenny, 2002).

6.1.6 Design best practice and mitigation techniques

Cooling water intake location

Earlier sections of the report dealt with engineering and plant performance factors, such as recirculation and air entrainment that influence CW intake location. Here, we look at how intake location should take account of biological factors in the local area. Biologically-informed planning not only helps to reduce impacts upon fish, crustaceans and other valuable biota; it also minimises risk of periodic inundations of fish and weed that can otherwise lead to serious operational problems and potentially large revenue losses. Selecting an intake location is an iterative process that takes account of all the above factors. Below we outline some biological considerations.

Fish-screening and fish-return techniques available today greatly reduce the risk of fish losses to impingement, whereas entrainment of early life-stages is harder to control. Therefore, biological considerations should focus more on avoiding entrainment risk than impingement.

A key reference that should be used alongside this section is the Environment Agency's Best Practice Guide for intake and outfall fish screening (Turnpenny and O'Keeffe, 2005). The Guide contains a great deal of information on environmental mitigation techniques for intakes, as well as background data.

Locality of power station

Generally, it is desirable to avoid construction in areas of notable aquatic habitat conservation or ecological value. These might include, for example, Natura 2000 sites[23], important fish spawning and nursery grounds, ecologically sensitive habitats, economically important shellfisheries and fish migration routes. Owing to the UK Government's rules on site selection for new nuclear build, there is little scope for moving the base locations of proposed developments and therefore siting questions must revolve around whether the best decisions are being made within the geographic planning constraints and whether the residual impacts after mitigation are acceptable or compensatable. Given that CW inlet and discharge tunnels can practically extend to a few kilometres in length, and that the proposed sites for new nuclear power stations

[23] Natura 2000 sites are sites identified as of Community Importance under Habitats Directive 92/43/EEC or classified as Special Protection Areas (SPAs) under Birds Directive 79/409/EEC

have sufficient land-area to build cooling towers etc, there nevertheless remains scope for minimising or avoiding impacts upon important habitats.

It should be noted that an Appraisal of Sustainability (incorporating the requirements of the Strategic Environmental Assessment (SEA) Directive 2001/42/EC) and a Habitats Regulations Assessment are being completed alongside development of the Nuclear National Policy Statement (NPS). These plan-level assessments inform the NPS and should address some of the issues above.

Shoreline, mid-channel or offshore options

Key considerations here will be the relative importance of different habitats represented in these zones.

In any coastal or estuarine situation, drawing a transect from the shoreline into deeper water will reveal strong biological zonation. Intertidal and saltmarsh areas are often the most highly productive, and are also important habitat for juvenile fish. Abstraction from these areas will increase the risk of drawing in juveniles. An example of this is seen at Fawley Power Station, Hampshire, which has an onshore intake connected to the deep shipping channel of Southampton water via a 500 m dredged channel. The adjacent saltmarshes drain into dredged channel as the tide falls, concentrating fish in the CW stream and increasing entrapment risk (Turnpenny and Utting, 1981). Had it been recognised as an issue at the time of construction, such an arrangement might have been avoided by locating the intake offshore via a pipeline, or by sheet piling the intake channel edges to ensure that the saltmarshes drained away from the dredged intake channel (Figure 6-12). A similar risk may also arise with an intake that opens near the low water mark, where fish in the intertidal tend to concentrate as the tide ebbs.



Figure 6-12 Fawley Power Station CW intake at low tide, showing the onshore CW inlet channel and bordering saltmarsh areas

Cefas and its predecessors in MAFF DFR have for many years conducted Young Fish Surveys around the coasts of England and Wales (Riley *et al.* 1986; Rogers *et al.* 1998). These used two standard scientific fishing techniques, a two-metre beam trawl and Riley 1.5-m push-net to sample coastal habitats starting at the surf zone and

extending into deeper waters to around 20 m. Data show zonation, particularly of flatfish species, zero-group concentrations being highest in the surf zone, giving way to 1+ and 2+ fish in deeper water (Figure 6-13). Data from these surveys for the east coast were a key factor in the choice of CW intake location for Sizewell B, where intake was pushed 600 m offshore (compared to 300 m offshore for the A station) to reduce the risk of drawing in weakly swimming zero-group fish (Turnpenny and Taylor, 2000).

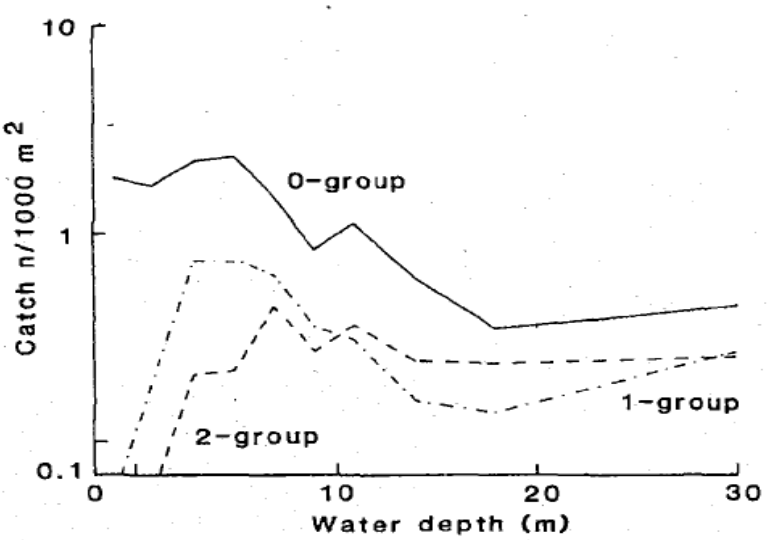


Figure 6-13 Cefas Young Fish Survey data showing zonation of sole (*Solea solea)* relative to water depth

Siting intakes within narrow estuaries brings a different problem. While locating the CW intake offshore reduces the risk of juvenile fish entrapment, it may increase the risk for any migratory fish such as salmonid smolts that migrate in mid-channel.

Given the potential for entrainment of early life-stages, intake siting should take particular account of spawning and nursery areas. Generally these locations are well known and mapped and should be shown on any GIS maps developed for the project. More important commercial ones are shown on UK DMAP[24] and can be found for example on UKOOA Fisheries sensitivity maps[25] and other marine GIS resources. Such sources may not include spawning grounds that are locally rather than nationally important. For example, a herring spawning ground at Beggar's Reach in Milford Haven (South Wales), which supports a spawning stock of 10-20 million adults, is of no significance to UK herring landings but may be an important ecological feature and contributes occasionally to the local inshore fishing industry (Clarke and King,1985).

When positioning of an intake occurs within a few kilometres of an inshore spawning ground, current patterns need to be taken into account. This applies to estuaries and the open coast, as tidal excursions will carry early life-stages this sort of distance on a single tide. Dispersion of larval herring from the River Blackwater herring stock, which is a Thames spring-spawning sub-stock, was investigated by the CEGB as part of the 1980s Bradwell B programme (Henderson and Cartwright, 1980). A slightly different modelling approach was later applied to the same stock by Fox and Aldridge (2000). Herring are benthic spawners, with adhesive eggs that attach to gravel beds until hatching. Once released into the water column, the larval herring disperse by a process of diffusion and current-driven drift. Samples were collected at various points down-

[24] Available as a download from http://www.bodc.ac.uk/products/bodc_products/ukdmap/ (accessed 17/02/09)
[25] Available as a download from http://www.cefas.co.uk/publications/miscellaneous-publications/fisheries-sensitivity-maps.aspx (accessed 17/02/09)

tide towards the proposed intake location, which was 12 km from the Eagle Bank spawning ground. Both studies reached similar estimates of between 18 and 25 per cent of larvae being washed past the section of channel in which the intake was to be located over the 70 days following hatching (though only a proportion of these would be entrained). This type of approach is more useful for benthic spawning species as the spawning areas are more discrete than those of pelagic spawners such as sole, plaice or bass. Other benthic spawners include, for example, shads (*Alosa* spp.) and sand smelt (*Atherina boyerii*).

Recent investigations have been undertaken as part of the environmental assessment process for the planned Pembroke CCGT power station, whose CW intake will be located about 10 km from the Beggar's Reach herring spawning ground mentioned above. Ichthyoplankton surveys close to the intake indicate that numbers of larval and post-larval herring in the CW volume required to supply the station during the post-hatching period would amount to no more than around 0.1 per cent of Milford Haven stock when expressed as adult equivalents.

The biological consequences of choice of intake opening depth have seldom been considered in past intake designs. As discussed in Section 5, the engineering preference is for deeper openings to avoid warmer surface layers and vortex formation. Fortuitously this is probably the better position for avoiding fish entrapment, as the bulk of fish drawn in are commonly pelagic species that favour the mid-to-upper water column. Bottom openings, conversely, will favour demersal and benthic species and the best arrangement may in most cases be to have openings a meter or so off the bed. This aspect needs to be considered on a site-specific basis according to conservation priorities.

Intake siting should also take account of the risk of operationally harmful inundations of biota. Little can be done to avoid exposure to mass inundations of jellyfish or sprat shoals in areas where these reach nuisance levels, as they are ubiquitous on these occasions. The emphasis in these cases should be on designing screening or deflection systems that can cope. Seaweed inundations, on the other hand, usually occur following storms, particularly after autumn dieback. Hydrographic factors then cause the material to accumulate in backwater areas or depressions in the seabed. Avoiding these areas when locating the intake can prevent later problems.

Biota exclusion and deflection techniques

Exclusion of fish, shrimps, weed and other organisms at the point of entry from the source water is greatly preferable to screening and returning them once they have entered the cooling system. Positive exclusion by fine mechanical screens is one approach but as discussed in Section 5.5 is unlikely to be considered for new nuclear sites. Other techniques depend on physical processes such as air bubble curtains to divert biota or upon behavioural stimuli to drive fish away from inlets.

Air bubble curtains

Bubble curtains are formed when a porous or perforated pipe is affixed to the seabed and fed with compressed air. The rising curtain of bubbles so formed has a variety of useful properties in CW intake applications, including the ability to deflect organisms and trash and to reduce the risk of surface oil entering and coating the heat exchangers following a spill.

The mechanism by which this occurs is illustrated in Figure 6-14. In the case of organisms in the water column, as they are drawn towards the bubble curtain, they can become entrained into the vertical convection current generated by the rising bubble

plume and brought to the surface. This process is probably also partly due to attachment by surface tension of bubbles to the organism, increasing buoyancy. Once at the surface, a surface current generated by the upwelling plume repels organisms away from the bubble curtain. By laying the bubble curtain diagonally with respect to tidal flow direction, current vectors perpendicular to the barrier line deflect organisms to one side. Figure 6-15 shows the concept in plan view for an offshore intake.

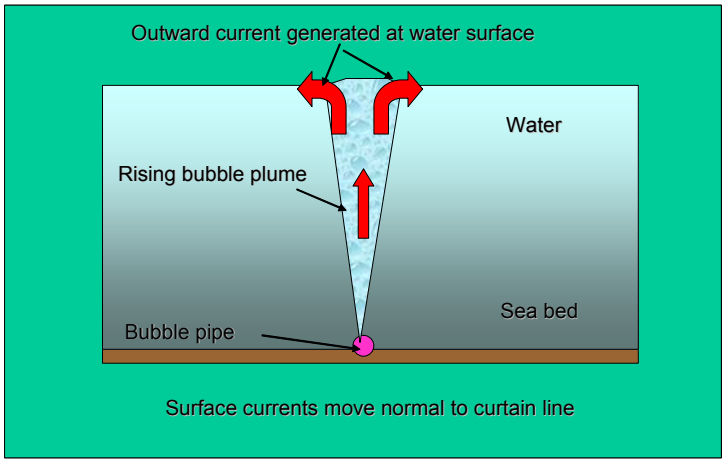


Figure 6-14 Schematic showing rising bubble plume and surface currents normal to the barrier line in a bubble curtain

A bubble curtain was installed during the 1990s across the combined CW intake entrances of Heysham A & B nuclear stations in Heysham Harbour (Turnpenny, 1993) and is still in use. The main intention was to reduce fish ingress at a time prior to the development of more modern methods (see below). The curtain extended across a diagonal line such that fish would be diverted away from the inlet and back into the harbour. Comparison of drum screen catches for alternating on-off periods showed that fish catch was reduced by about one-third. An unexpected benefit was that shrimp catches were reduced by around two-thirds, presumably as a result of the purely physical processes described above.

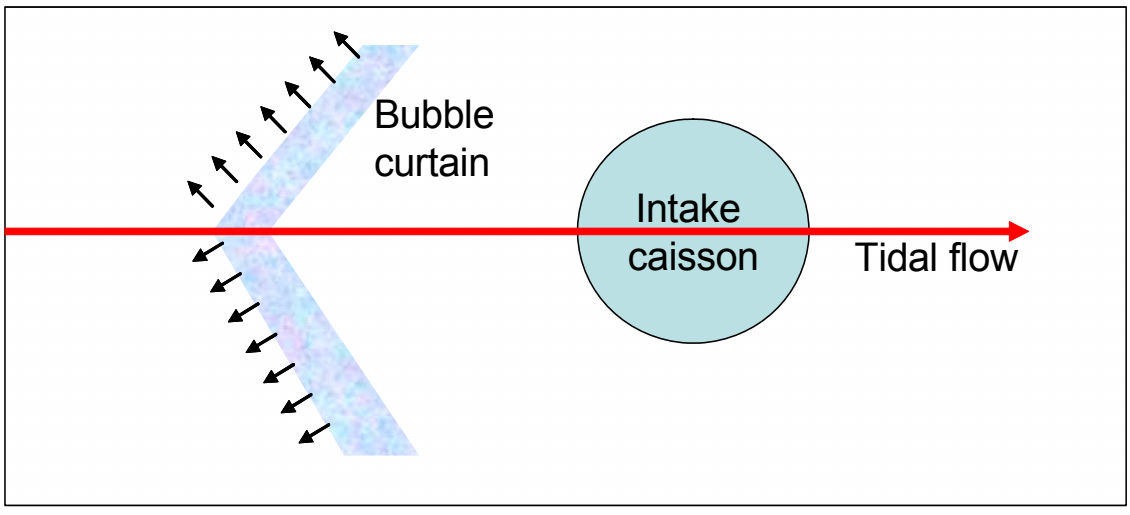


Figure 6-15 Schematic showing bubble curtain deflection concept for an offshore intake

While newer methods can offer better levels of protection for fish, the merits of the bubble intake caisson operating in a tidal flow curtain for deflection of invertebrates and against oil ingress in port and harbour locations should not be ignored. The concept

drawing in Figure 6-15 is based on a proposed scheme by Fawley Aquatic Research (now Jacobs Engineering) to reduce jellyfish ingress at Hunterston B nuclear station. The idea is to raise jellyfish in the water column by air becoming trapped beneath the umbrella, taking them over the top of the submerged intake structure, the chevron shape of the curtain at the same time pushing them outwards, away from the intake. It has not been implemented to date.

Bubble curtains have also been used to reduce siltation risk at some power stations, where accretion can partially block openings and cause intake velocities to exceed licensed conditions. An experimental curtain was used to dislodge silt depots around Barking power station on the Thames Tideway but was found to be difficult to maintain. Another is used across the CW intake channel of Fawley power station to reduce sedimentation across the inlet at times when the plant is not drawing CW flow.

Velocity control

Active or behavioural deflection of fish first requires velocities at the intake entrance (often known as "approach velocities") to be low enough for fish to avoid. Design criteria for fish conserving intakes therefore rely on good knowledge of the swimming performance of different fish species and life-stages. As fish are poikilotherms (body temperature is determined by external water temperature), water temperature has a strong influence on metabolic rates and therefore swimming performance.

At one time, relatively crude swimming-speed criteria based on non-indigenous fish species were used for this purpose. More recently the power and water industries, as well as the Environment Agency, have seen the need to develop a database for UK inland and saltwater species. Relevant information is summarised in the Best Practice Guide (Turnpenny and O'Keeffe, 2005). For most power plant intake purposes a design fish-escape velocity of 0.3 ms^{-1} will be suitable and meet best practice requirements. Where a different value might be preferable, the guide should be consulted.

Two further design issues relating to velocity are relevant to offshore intake structures. One concerns elimination of vertical velocity components, which fish are ill-equipped to resist. The problem can be overcome by fitting a "velocity cap" over the intake to ensure that water enters horizontally. Care should be taken to ensure that the aspect ratio (height:length) of the entrance meets the criteria presented by Schuler and Larson (1975) (Figure 6-16), which ensure that vertical components are eliminated. Velocity caps are a powerful form of mitigation that can reduce entrapment by up to 90 per cent, and are already used in some newer UK offshore intake designs (such as Sizewell B: Turnpenny and Taylor, 2000).

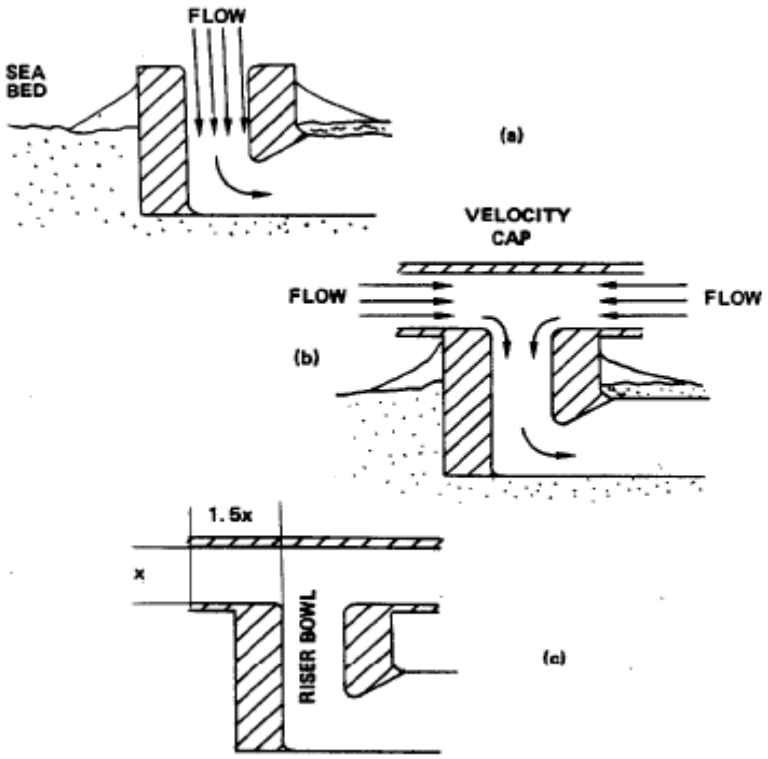


Figure 6-16 Velocity cap: (a) section of uncapped intake showing vertical draw-down pattern, (b) section of capped intake showing horizontal flow pattern, (c) as (b) but showing critical relationship between vertical opening [x] and length of horizontal entrance [1.5x] for fish reactions (after Schuler and Larson, 1975)

The second issue concerns the orientation of intake openings in relation to tidal or river flow. Many UK stations with offshore intakes can take water from 360^{o} around the circumference of the structure. Hydraulic modelling studies have shown that resulting velocities on the upstream side of the intake approximate the sum of the intake velocity (calculated as if there were no crossflow) and the approaching tidal stream velocity (Turnpenny, 1988a). Whereas Sizewell B was designed with a nominal entrance velocity of 0.5 ms^{-1} under static water conditions, the actual approach velocity can be several times this value in mid-flood and mid-ebb periods, leading to higher fish impingement rates at these times. The Best Practice Guide (Turnpenny and O'Keeffe, 2005) presents a conceptual design (Figure 6-17), the low velocity side-entry (LVSE) intake that appears from initial modelling tests to overcome the problem and maintain constant velocities around the tidal cycle. As yet, this design has not been built and requires further model testing to develop the detailed design and ensure its suitability.

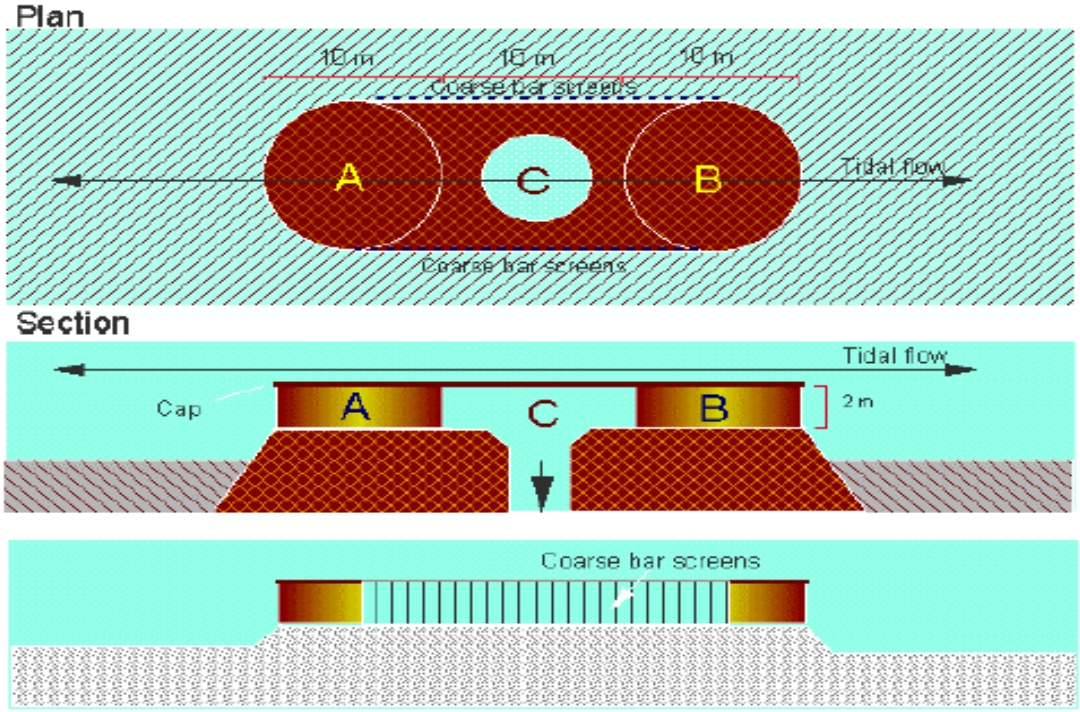


Figure 6-17 Concept design for a low-velocity, side-entry offshore intake structure with velocity cap, based on physical and hydraulic model tests carried out at Fawley Aquatic Research Laboratories (Turnpenny, 1988a and unpublished). A and B are the cylindrical caissons. Water enters through C.

Acoustic fish deterrents and strobe lights

A recent innovation to reduce fish entry into CW intakes is the acoustic fish deterrent (AFD) system. AFDs were originally investigated by the CEGB for Severn Barrage tidal power generation, where the need to exclude fish from hydropower turbines passing up to 200 m^3s^{-1} was considered to preclude conventional physical screening methods. When the Severn Barrage project of the 1990s was dropped, the nuclear industry saw the benefits of pursuing AFD development and early trials were carried out at Hinkley Point (Turnpenny, Thatcher and Wood, 1994) and Hartlepool (Turnpenny *et al.* 1995) nuclear stations. Hinkley Point trials showed that fish impingement rates could be affected by the AFD's sound field, but catches increased rather than decreased! The AFD system used at Hinkley was made up of off-the-shelf military hardware and was not ideally suited to generating frequencies audible to fish. The reason for increased fish catch appeared to be the sound field causing fish to concentrate at the depths from which water was abstracted. For the second series of trials, conducted at Hartlepool, more suitable acoustic transducers ("sound projectors") were developed. These, along with better use of acoustic modelling software to optimise the sound field, led to the successful deflection of a large proportion of fish. Following privatisation of the CEGB and the nuclear industry, independent private sector development continued the refinement of AFDs, there now being a variety of AFD systems suited to different uses.

AFD systems have become commonplace at large abstractions where physical fish screening (as opposed to trash screening) is problematic. They have been installed at large thermal stations such as Doel nuclear station (Belgium), Fawley (oil-fired, Southampton Water), Shoreham (CCGT, West Sussex), Great Yarmouth (CCGT, Norfolk), Marchwood (CCGT, Southampton Water) and Staythorpe (CCGT, R. Trent) and systems are scheduled for installation at other new stations. A system was recently

installed at Lambton Generating Station, a large direct-cooled plant on St Clair River in Canada.

The fish deflection performance has been investigated at a number of installations. Testing usually involves comparing screen catches for a sequence of AFD “off” versus “on” days. The cases most representative of the latest technology combined with use of acoustic modelling techniques are Doel (table 6-6) and Lambton. The effectiveness of the system depends mainly on the hearing sensitivity of individual species, although other factors such a swimming ability must be taken into account. Most fish possess a swimbladder, which increases sensitivity to sound pressure. Such species (which include many demersal species, including most members of the cod family, bass and other ‘roundfish’) can be deflected with sound. AFD deflection efficiencies are usually between 50-70 per cent for these species. A smaller group of species, known as ‘hearing specialists’, have anatomical adaptations that increase sensitivity to sound. Examples include pelagic fish such as herring and sprat, but also members of the carp family and catfishes. Deflection efficiencies are correspondingly higher, around 80-95 per cent, for these species. A third group comprises flatfishes and other benthic species with reduced or no swimbladder function. Efficiencies are considerably lower for these species. In theory, installing more sound projectors and increasing sound levels will increase the efficiency for all species but the law of diminishing returns applies, and it is normally more cost-effective to add a second type of stimulus, such as strobe lights, and/or to combine AFDs with a fish recovery and return system.

AFD systems that have been used most successfully at power stations deliver sound signals in the 10 to 3,000 Hz hearing band that is audible to most fish[26]. Systems comprise a single signal generator, designed to provide a suite of signals selected according to species, a bank of audio power amplifiers, and a sound projector connected to each amplifier (Figure 6-18). Each line can also be provided with a diagnostics unit, which allows the performance status of all elements of the system to be monitored remotely, for example in the plant control room or via a website.

Sound projectors (Figure 6-19) are installed in a matrix or array across intake entrances, creating a localised, repellent sound field. Normally these are attached to vertical rails incorporating a winch mechanism that enables them to be drawn to the surface for maintenance (at least annually). Figure 6-19 shows the substantial sound projector support structure used at the Canadian Lambton station, designed to ensure that the sound field extends to a point where velocities are low enough for fish to escape. An important consideration for nuclear stations under construction, especially those with offshore CW intakes, is to ensure that suitable cable ducts and cable runs are provided between the onshore station buildings and the intake structure. As cables are difficult to replace, it is also good practice to provide additional spare cabling to allow for future expansion of the AFD system, if necessary, or to replace failed cables.

[26] Ultrasound (>100 kHz) systems have been used with some success in the USA for certain species with ultrasound sensitivity (clupeid family) (Turnpenny and O’Keeffe, 2005) but are not effective for other species and would not therefore be suitable for new UK nuclear build.

Table 6-6 Deflection efficiencies reported for the acoustic fish deflection system at Doel nuclear station (Maes *et al*. 2004)

Fish species	Deflection efficiency ('on' versus 'off')	Statistical significance
Herring (*Clupea harengus*)	95%	P<0.001
Sprat (*Spratus sprattus*)	88%	P<0.001
Smelt (*Osmerus eperlanus*)	64%	P=0.004
Bass (*Dicentrarchus labrax*)	76%	P<0.001
Flounder (*Platichthys flesus*)	46%	P=.0.028
Gobies (*Pomatoschistus* spp.)	50%	P>0.05 (NS)

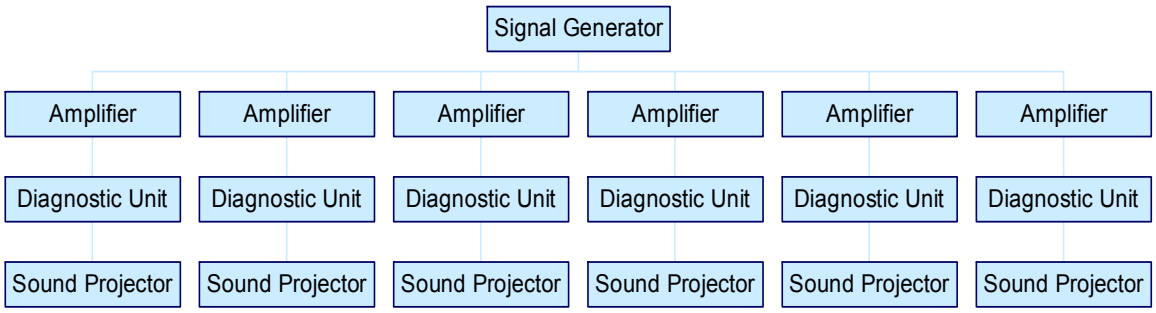


Figure 6-18 Schematic layout of a sound projector array (SPA) acoustic fish deterrent system with six sound projectors. Large systems may use up to 60 sound projectors, each with a dedicated amplifier and diagnostics unit.



Figure 6-19 Example of an acoustic fish deterrent support structure constructed across a shoreline CW intake at the Lambton Generating Station, St Clairs River, Canada. The vertical beams each act as rails, which allow sound projectors to be raised and lowered for maintenance. The positioning of the structure ahead of the intake opening ensures that the fish repulsion zone is in an area of low water velocities from which the fish can escape. The inset shows an individual sound projector. (Main photo courtesy Paul Patrick, Kinectrics).

Strobe lights have been used for many years to deter fish, but have been limited by the high voltage requirements of traditional xenon strobes (kilovolts) and their restricted bulb life. The recent development of low-voltage (under 12 V) strobes using high-intensity light-emitting diodes (LEDs) has overcome these problems and the abstraction licence for the proposed Pembroke CCGT requires strobes to be added to the AFD system on an experimental basis. Strobes appear to be one of the few effective means of deterring eels, although they are also expected to augment the AFD deflection efficiency for other fish species. More information on strobe and AFD systems can be found in the Best Practice Guide (Turnpenny and O'Keeffe, 2005).

Biota recovery and return techniques

Fish recovery and return (FRR) systems derive from standard band and drum screen designs but incorporate modifications that reduce the risk of fish injury (see Section 6.1.3 for a description of stressors). Key features are:

- replacement of standard trash elevators with water-retaining fish buckets;
- continuous screen rotation so that fish are not impinged against the screens for long periods before removal;
- use of low-pressure backwash sprays for fish removal (usually followed by high-pressure sprays to clear more persistent fouling);
- ensuring backwash launders are smooth, free from potential snags and flushed with copious quantities of water (Figure 6-20);
- a fish return pipe or launder to put fish back to the source water body at a point where they are unlikely to be returned to the intake point. The fish return line should enter the water below the lowest astronomical tide mark.



Figure 6-20 Example of fish return launders. Launders should be covered to reduce predation risk. Larger radius (3 m) swept bends reduce the risk of debris and fish becoming caught in bends.

Research has shown that the design of the fish buckets is critical, if fish are not to be flushed out of it by the shearing effect of water being pumped through the screen mesh (Figure 6-21a). The problem is avoided by using buckets of a suitable hydraulic design (Figure 6-221b), recently fitted, for example, at the new Marchwood CCGT station.

A new issue in the design of fish buckets is the need to handle large, sinuous species such as adult eels and lampreys. Survival rates of eels in fish return studies have generally been low (see Clough *et al.* 2003). The main reason is that eels writhe and fall out of the buckets, probably many times before being removed by the screens. As a result they become exhausted and often have multiple wounds and sores by the time they are removed.

The same may be true of lampreys, although observations are fewer. Fish buckets typically have an opening width of 60 mm (Figure 6-22) which may be too small to handle these species safely. Given the recent EC Eel Regulation and the conservation status of lampreys under the Habitats Directive, there is an urgent need to investigate this issue and develop suitable design criteria.

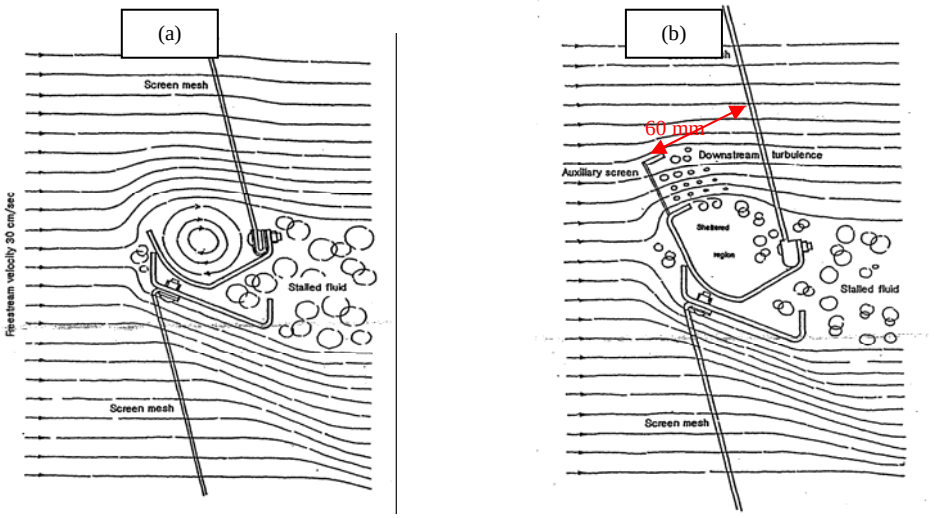


Figure 6-21 Streamlines around a travelling screen fish bucket (a) traditional, showing rotational flow and (b) modified to stall flow within bucket (Fletcher *et al.* 1988). Width of bucket opening is typically 60 mm in standard designs.

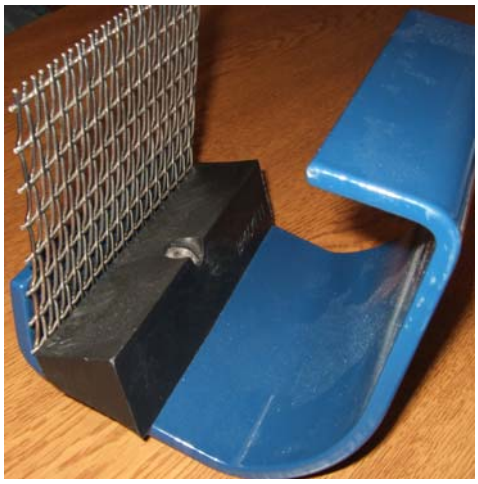


Figure 6-22 Example of a fish bucket profile. The mesh to the left represents part of one of the main screen panels (Eimco Water Technologies).

Biocides used for biofouling control within the CW system should ideally be injected downstream of the screenwell to avoid toxic stress to the fish being returned. While Turnpenny (1992) concluded that brief exposure to TRO may be tolerated by fish, exposure to sublethal effects of TRO may disorientate fish and increase the risk of predation once they are released back to the wild. The Best Practice Guide (Turnpenny and O'Keeffe, 2005) therefore recommends that exposure be avoided, although screenwells are often chlorinated. This recommendation has yet to meet with a satisfactory response from developers, who are concerned about biofouling accreting within the screenwells and potentially blocking screens, a problem which may need to be addressed with intermittent shock dosing or use of antifouling coatings.

The Best Practice Guide gives the following summary of design criteria for FRR systems, which should be applied as far as possible at new nuclear stations:

- The design of the fish buckets should be optimized for fish handling.
- Screen meshes should be smooth and fish-friendly, constructed from woven stainless steel or plastic mesh.
- Mesh size should be 6 mm or less.
- Low-pressure backwash sprays (under one bar) should be used for fish removal; higher pressure jets may be used at a later point in the cycle to wash off debris.
- The geometry of the collecting hoppers should ensure that fish washed off the screens cannot fall back into the screenwell.
- Biocides should be applied only downstream of the screens unless it can be shown that the toxic risk is negligible.
- Fish return gullies should be smooth, with any joints properly grouted and finished. They should have a minimum of 0.3 m diameter with at least 0.5 m diameter or larger for the main return channel run (above 30 m).
- Fish return lines should be covered to prevent bird predation and algal growth, with access for cleaning.
- Swept bends of radius greater than three metres should be used.
- A dedicated fish return line should be provided.
- A continuous wash-water supply should be provided to ensure sufficient depth to keep fish immersed and moving along the return line.

Putting these guidelines into practice at recent new-build stations has already revealed further details that should be specified, and other requirements may come to light as experience is gained. The following additional design criteria should be considered:

- In order to minimise the fish handling time the screens should be rotated at a constant speed of at least 1.5 m per minute.
- Any changes in slope of launders should use a minimum 3-m swept bend radius for vertical bends to avoid flow separation from the launder bed.
- The fall on launder sections feeding into horizontal bends should be restricted, as accelerating flow may cause standing waves and overtopping in the bends. This may be accomplished, for example, by restricting the fall on launder sections upstream of horizontal bend to 1 maximum of 1:50.

- Turbulence should be minimised to reduce the risks of fish exhaustion and injury. It is recommended that energy dissipation throughout the system should be kept at or below 100 Wm^{-3}. This particularly applies to any fish sampling or holding facility that may be incorporated for fish impingement monitoring purposes. The method of calculating energy dissipation is described in the Environment Agency Fish Pass Manual (Armstrong *et al.*, 2004).
- While most FRR systems allow the high-pressure backwash (primarily intended to remove material sticking to the screen) to discharge to a trash basket, provision should also be made to allow discharge of backwashings via the fish return launders. This facility can be used in cases where significant quantities of shrimps, for example, would otherwise be sent to landfill rather than returned to sea. At times when it is mostly weed and rubbish that are removed at the high-pressure wash stage, the material could be collected for disposal.

The geometry of band screens makes them more adaptable than drum screens to fish handling and most FRR development has been based on the band screen format. As it is likely that drum screens will be used in new nuclear stations, the difficulties of drum screen geometry will require attention. The main issue is that the tipping radius of the fish bucket is much larger than on band screens, so fish tend to fall out of the bucket towards the collection hopper over a relatively large arc. Thus, a large proportion of fish can fall back into the screenwell and may have to be handled multiple times before successful removal. Figure 6-23 illustrates this problem with a photograph from one nuclear plant where scaffold boards had been placed in front of the hopper and were seen to collect quantities of fish that missed the hopper.



Figure 6-23 Fish that have overshot the hopper collecting on a scaffold boards inside a drum screen chamber

Where the use of biocides upstream of fish return facilities is critical to security of plant operation, a suitable ecotoxicological risk assessment should be carried out to ensure that any risk of adverse lethal and sublethal effects on fish is minimised, and that the

level of ecological impact associated with biocide use is acceptable. Wherever practicable, biocide application should be conducted on inactive legs of the CW system (for example, soak application when no abstraction is taking place) or at times of the year when fish entry is likely to be minimal.

Neither the AFD nor FRR systems used in Britain are capable of protecting small fish of less than around 30 mm in length, which are vulnerable to entrainment.

6.1.7 Special measures at Natura 2000 sites?

No separate regulations or standards on impingement and entrainment apply to Natura 2000 sites but particular attention needs to be paid to species listed in the conservation objectives for the site. Fish species listed under Annex 2 of the Habitats Directive (92/43/EEC) include:

- River lamprey (*Lampetra fluviatilis*)
- Sea lamprey (*Petromyzon marinus*)
- Brook lamprey (*Lampetra planeri*)
- Atlantic salmon (*Salmo salar*)
- Bullhead (*Cottus gobio*)
- Spined loach (*Cobitis taenia*)
- Allis shad (*Alosa alosa*)
- Twaite shad (*Alosa fallax*).

Of these, the three lamprey species as well as loach and bullhead are benthic and likely to be insensitive to AFD techniques but amenable to FRR methods. Conversely, salmon and shad are too delicate for FRR but can be deflected by behavioural methods.

Compliance with the Habitats Directive does not necessarily mean that the project must lie within the designated area of the SAC; if the project might indirectly have an impact on the SAC, such as an abstraction on the migration path of a designated species attempting to reach the SAC, then it will also be subject to the Habitat Regulations[27].

Any development that might affect a Natura 2000 site will be subject to a regulatory assessment process by the competent authority (The Environment Agency of England and Wales). In such an assessment there will be an initial *screening* stage, which seeks to determine whether the proposed development is likely to have a significant impact on conservation features of the European-protected site. Where there is considered to be a significant risk, then the competent authority will undertake an appropriate assessment to evaluate that risk. Where a significant impact is predicted, approval to proceed with the development will be granted only if the proposed project can be modified to mitigate the significant impacts, or is subject to 'overriding public interest' considerations. Further details can be found in various guidance documents (see e.g. http://www.mceu.gov.uk/MCEU_LOCAL/Ref-Docs/EN-HabsRegs-AA.pdf [accessed 08/01/10).

[27] Conservation (Natural Habitats & Conservation) Regulations 1994

6.1.8 Residual impacts

The environmental impact assessment for a new nuclear station should attempt to quantify, among other things:

(a) the proportion of fish present in the approaching CW flow that would be deflected by mitigation measures at the intake structure;

(b) the proportion of impinged fish that would be returned safely to the source water body by FRR measures;

(c) the proportion of entrained early life-stages that would pass back unharmed to the source water body.

Residual impacts will then arise from impinged and entrained fish that are returned to the wild, dead or injured. The residual impact is reduced further by the reassimilation of this biomass back into the food chain, although this is more difficult to quantify.

Residual impacts from a development may be offset by appropriate forms of compensation. These should be in kind, and at least commensurate with the level of residual impact. The California Energy Commission approach provides a model for this (CEC, 2005), in which the residual biomass loss is converted to habitat production foregone (HPF) (Section 6.1.5) and then new habitat to the equivalent or greater area is created. UK examples of best practice for lost estuarine and coastal production are emerging from parallel considerations in various industrial sectors where coastal habitat projects have been funded (Table 6-7).

Table 6-7 Example of large-scale habitat compensation projects in the UK (Scottish Parliament, 2008)

Existing large scale habit projects in the UK	
South Wales-Newport Wetlands Reserve	• 439ha site as compensation for habitat lost to Cardiff Bay Barrage • Countryside Council for Wales, Newport City Council & RSPB Partnership • 50,000+ visitors expected p.a.
Humberside–Alkburgh Flats	• 370ha of intertidal and other habitats created by the Environment Agency • £6.1m project resulting in £23.6m environmental and flood defence benefits
South Yorkshire-Dearne Valley	• restoration of network of sites to create high quality natural habitats • significantly improving area's image and appeal to new residents, businesses and visitors
Teesside-Saltholme	• 380ha former industrial site • will open as RSPB reserve in 2009 with 100,000 visitors p.a. expected • aims to help attract highly skilled, creative individuals to the area • substantial funding from Regional Development Agency - One North East
South East England-Wallasea Island	• RSPB managed project • restoration of 620ha of land back to coastal marshland • adjacent to 115ha DEFRA funded port-development compensation project
Thames Gateway-Rainham	• 353ha RSPB reserve • former MOD firing range • major greenspace for the Thames Gateway

An important consideration in determining compensation is that it should match as closely as possible the loss of species and life-stages. It is normal practice to provide in compensation a larger habitat area, around 1.5 times the estimated HPF figure, to allow for error in the estimate, any underperformance of the replacement habitat and habitat maturation time.

7 Conclusions and recommendations

7.1 Environmental effects of cooling systems

7.1.1 Impingement

Fish impingement is minimal at inland sites with wet-tower cooling but can account for tens or even hundreds of tonnes of fish per annum at estuarine and coastal sites with direct cooling. Quantities of fish involved depend on the size and CW flow of the plant, proximity of intake to fish habitat and concentrations of fish and mitigation measures installed. The largest quantities recorded tend to comprise mainly pelagic fish such as sprat. At Sizewell A station, of 36 tonnes of fish impingement recorded in one year, 33 tonnes comprised sprat alone (Turnpenny *et al.* 1989). On occasions, sprat inundations at East Coast sites have amounted to hundreds of tonnes in a single day.

Although, these quantities appear large, they are small in relation to the industrial fisheries for sprat and of more interest are losses associated with impingement of juveniles of species of commercial or conservation interest. The removal of juveniles (undersized catch) by power stations has always been poorly regarded by the fishing industry and the Sizewell studies showed that for every tonne of juveniles removed by impingement, around fourteen tonnes of adults were in effect lost from production.

Mitigation techniques to reduce losses to impingement have progressed considerably in the past decade. At Shoreham CCGT station, which is fitted with an AFD system, the annual impingement rate averaged 3.8 kg per $10^6 m^3$, the lowest rate recorded for any UK station (see Figure 6-5). Combinations of techniques, including use of velocity caps (offshore intakes), AFD and FRR systems and other technologies that will be considered for the next generation of stations (LVSE intake designs, strobe light deterrents) should further reduce losses to impingement.

7.1.2 Entrainment

Entrainment affects all types of plankton but impacts on permanent zooplankton and phytoplankton (holoplankton) appear to be minimal in most cases, owing to the rapid turnover time of plankton populations.

Ichthyoplankton entrainment is potentially a more serious concern and has been extensively measured at US power plants under CWA s.316(b) rules, which require plant operators to assess entrainment impacts and put in place measures to reduce impacts by 60-90 per cent of unmitigated values[34].

Fish entrainment has been less thoroughly investigated than impingement at UK and European sites. The scarcity of entrainment data reflects a lack of regulatory interest in this less obvious form of impact. Entrainment impacts are potentially more significant than those associated with impingement, particularly now that impingement impacts can be largely mitigated.

[34] See e.g. http://www.powermag.com/water/Alternative-cooling-water-intake-analysis-under-CWA-Section-316b_1045.html (viewed 24/02/09).

The level of fish entrainment risk is determined by proximity to spawning grounds, and avoiding these and areas where fish eggs and larvae concentrate when locating the intake are the main means of minimising entrainment impacts.

The assumption of 100 per cent mortality of ichthyoplankton during cooling system passage advised by US EPA is not supported by UK power industry studies in which survivorship of entrainment has been measured using the EMU cooling system simulator. Assumed high mortalities associated with entrainment are a key factor in US EPA policy to limit direct cooling. However, UK data on survivorship are presently limited to a few species/lifestages. There is now a need for the industry to provide more comprehensive data covering vulnerable lifestages of species of commercial or conservation importance. It will also be necessary to demonstrate that the conditions tested effectively emulate those of new nuclear stations, which may entail conditions that differ from those already tested (based on the Sizewell B CW system profile).

7.1.3 Thermal discharges

Thermal discharge effects can be separated into near-field effects associated with the thermal plume and wider-range effects associated with the long-term heat field.

Water temperatures within the plume itself will typically be near or at the discharge temperature, which could be up to 10^{o}C above ambient, depending on generating load and permitted ΔT value. Warm-water species may benefit from the plume and mobile species such as bass may move into the plume and benefit from food supplies, leading to faster growth and extended growing season. Shads and lampreys are also of Lusitanian origin (Figure 6-26) and are likely to benefit from warming in British waters. Conversely, some cold-water species will be eliminated from the swept area of the plume. These patterns are most evident in the extreme case of the Kingsnorth discharge, where the plume is confined within a tidal creek. This has provided excellent case studies of thermal effects (Section 6.3.6). Owing to its buoyancy, the warmer parts of the plume seldom have an impact on the benthos, except where the outfall is located onshore and warm water runs across the foreshore. Where this occurs, the plume may also cause scour until a stable bed form is created.

Where the plume impinges on the seabed in tidal waters, the discrete interface between the warmed effluent water and the cooler receiving water will move across the seabed: this rapid change in temperature represents the most severe thermal impact on the benthos, and will lead to a significant loss of species where the ΔT is above 3ºC. As littoral species are naturally more tolerant of thermal stress, the common result in the vicinity of thermal outflows is a down-shore movement (into the sublittoral) of the vertical zonation of benthic plants and animals, except where these are constrained by other environmental factors (such as light) when they are simply eliminated. These effects on the benthos are commonly restricted to within some 500 m of the outfall (depending on flow rate and volume, and physiography of the receiving water body).

A recent area of interest has been whether peripheral areas of the plume attaching to the foreshore of estuaries might inhibit longshore migrations of juvenile fish such as elver, flounder, goby, dace and smelt. New laboratory studies have shown that most of these species are indifferent to crossing temperature interfaces of up to a few degrees, with the possible exception of smelt. The smelt is a cold-water fish of the salmonid family and in the laboratory studies avoided ΔT values above +4^{o}C. Where temperatures are expected to exceed avoidance thresholds, it will be necessary to ensure that this only happens for a small proportion of the tidal cycle, allowing sufficient window of opportunity for fish to migrate past.

Similarly, plume behaviour and temperature values in estuarine channels or at the mouth of an estuary must not be allowed to impede the free movement of migratory fish such as salmon and shad that use the larger part of the channel cross-section. Again, salmonids, being cold-water species, are the most sensitive to such effects but are not impeded by ΔT values under $+3^{o}C$.

The long-term heat field can have a number of effects, for example shifting zoogeographical limits of species already close to their northern (warm-water species) or southern (cold-water species) distribution. This occurs on a wider scale in the context of global climate change (and has always been a natural factor affecting marine pelagic fisheries) but for power station discharges has been seen most clearly in bass stocks on the east coasts of England and Scotland, where warming has improved overwintering prospects. Another interesting example was the enhanced survival of oysters in the River Blackwater around the Bradwell outfall during the severe 1962-63 winter. The opposite side of the coin is that cold-water species such as salmon and smelt which presently live close to their upper limits in southern British waters, may eventually be displaced by rising temperatures associated with climate change; where the long-term heat field of a power station coincides with a confined stock of such species, their displacement can be expected to accelerate.

The long-term heat field can also influence seasonality, through causing temperature-triggered biological events to occur earlier or later in the season. Higher temperatures usually mean an extended growth and reproductive season for warm-water species but may mean shorter seasons for cold-water species forced to live at temperatures above optimum. These effects are within the normal range of inter-annual variation for most species and will only be significant for climatic fringe species.

Other water quality effects associated with thermal discharges mainly arise from biocide residues. Chlorine, produced by electro-chlorination plants or derived from sodium hypochlorite, is by far the most widely used antifouling product and is used to prevent build-up of macrofouling (mainly mussels) and condenser slimes. It is normally injected at the CW intake to achieve an effective concentration of around 0.2 mg l^{-1} at the condenser inlet and will have declined to a fraction of this value through a series of reactions that take place during cooling system passage, normally allowing discharge consents to be met without the need for any form of dechlorination. After discharge, the concentration rapidly falls further so that there is no acute toxicity to fish or other species in the receiving water. During further stages of reaction with seawater, the residuals form a series of chlorination byproducts (CBPs), mainly organohalogens, which bond to sediments or in some cases can bioaccumulate in the food chain.

Dissolved oxygen concentrations in water are bound to temperature, first through solubility effects and secondly through temperature coefficients of metabolic processes. In most cases, effects of thermal plumes and long-term field on DO concentration will be insignificant but this conclusion may not hold in sewage-polluted estuaries where higher temperatures will drive up biological and chemical oxygen demand. Where this risk is apparent, water-quality modelling will allow an assessment to be made.

7.1.4 Cooling tower effects

The greatest public concerns over cooling towers are the visual impacts of the plumes on the landscape. Planning authorities are therefore reluctant to approve large, natural draught cooling towers. Other cooling tower types have a lower visual profile but take up large areas of land and are less energy-efficient.

Vapour plumes from wet cooling towers represent an aesthetic impact and can cause local fog formation. Ice formation on the ground during cold weather appears to be limited to areas in close proximity to the tower bases and is usually confined within the

power station site. Similarly, salt deposition around seawater-cooled towers appears to be a minor localised effect, compared with natural salt deposition in coastal areas.

Legionnaires' disease, caused by the pathogen *Legionella pneumophila*, is the primary health concern associated with cooling towers. While outbreaks of Legionnaires' disease have been associated with air-conditioning towers, inevitably leading to concerns over the much larger cooling towers used on power stations, to date there has not been a single outbreak reported from this cause. Investigations have shown that, while the causative pathogen can exist within the lower parts of cooling towers, they have not been found above the eliminators or in the open air above cooling towers. However, it has not been proved that an outbreak could not occur. The risk is most likely to be associated with poor cooling tower maintenance (such as faulty eliminators) combined with atmospheric conditions favouring survival of the pathogen (dull, humid conditions).

7.1.5 Energy efficiency of different cooling systems

For reasons outlined in Chapter 2, wet (evaporative) cooling towers are more efficient in terms of delivering low recool temperatures, than dry towers and direct cooling is more efficient than tower cooling. Inter-station comparisons are of limited value since the major components and their design points will have been optimised for local conditions. However, it is now becoming standard practice to assess alternative cooling arrangements as part of the design and consenting process, comparing like with like. RWE NPower have compared direct cooling with two alternative methods for the steam turbines in the power-train of a notional 1000 MWe CCGT station (Mclauchlan, 2009). These data are show in Table 7-1.

The analysis in Table 7-1 was generated using modelling thermodynamic tools and selecting representative condenser back pressures for each plant design. It assumed a fixed boiler heat that would provide 1000MW net electrical output for a direct once-through cooling plant design. The Mechanical Draught and ACC designs were then run with this fixed heat input that therefore results in a lower gross generated electrical output due to the higher condenser back pressure. The total net electrical output reduction is a result of the lower generated gross output and auxiliary power requirements.

For the Mechanical Draught design, 78% of the net electrical output reduction is due to the lower gross generated output power with the remaining reduction coming from the increased auxiliary loading (combination of CW pumps and fan power). This is relative to the direct once-through cooling.

For the Air Cooled Condenser design 97.5% of the net electrical output reduction is due to the lower gross generated output power with the remaining reduction coming from the increased auxiliary loading (fan power). This is relative to the direct once through cooling and therefore highlights that the fan power is only slightly greater than cooling water pumps for the once through system.

Relative to the 1,000 MWe nominal rated output of the plant, these figures imply 2 to 2.3% lost electricity output.

Table 7-1 Potential loss of electrical output through the use of some alternative methods of cooling a 1,000 MWe power station, using direct cooling as the base case (Mclauchlan, 2009).

Parameter		Direct once-through cooling	Mechanical draught wet cooling tower	Closed-circuit air-cooled condensers
Condenser Design Pressure (mbar)		29	65	85
Reduction in electrical output (MW sent out)	MWe	0	42	47.4
	%	0	4.2	4.74
Reduction in Overall Cycle Net Thermal Efficiency (% points)		0	2	2.3

7.2 CW system design best practice

7.2.1 Selecting the most suitable system

General

Earlier sections of this report present the gamut of options available for power station cooling. Factors influencing the choice may include:

- sensitivity of source waters to abstraction impacts (entrainment and impingement), indirect cooling methods requiring less water and thus reducing these impacts;
- heat sink capacity of receiving water, lower capacities favouring indirect cooling methods;
- planning limitations on use of cooling towers (aesthetics, fog and so on);
- comparative lower thermal efficiencies of indirect cooling methods, therefore increasing carbon emissions per unit of electricity produced.

The first two of these favour indirect cooling, the second two, direct cooling.

Further factors that need to be considered are:

- whether predicted abstraction- and thermal-related impacts in the given situation exceed an acceptable level;
- whether the impact can be mitigated or compensated by way of, for example, replacement habitat.

Table 7-2 A comparison of cooling options

Environmental concern	Direct cooling	Cooling towers		
		Natural draught (wet)	**Mechanical draught (wet)**	**Natural draught (dry)***
Generation efficiency	High efficiency Uses less fuel so lower aerial emissions	Typically 0.5 - 1.5% less efficient than direct cooling	Typically ~2% less efficient than direct cooling	Lowest efficiency 2 - 3% less efficient than direct cooling
Complexity	Low	Moderate	High	Very high
Water abstraction	High	Moderate/low	Moderate/low	None
Abstraction effects	Site-specific -depends on characteristics of receiving waters			
Water consumption	None on-site	Moderate	Moderate	None
Visible plumes	None	Moderate	Moderate/low	None
Ground fog & icing	No icing. Local fog plume over shoreline discharges	None	Possible	None
Visual impact	Occasional foam or 'slick' at outfall	High	Moderate	High
Noise	None	Low	Moderate	Low/none
Discharge effects	Site-specific -depends on characteristics of receiving waters			
Waste disposal to landfill**	None if using fish recovery & return***	Moderate	Moderate	Moderate/none
Land use on-site****	None/low	Moderate/high	Moderate	High

* See sections 3.1.9 and 3.1.11

** Wastes from wet towers are mainly silt (non-hazardous); from dry towers, glycol (non-hazardous), if used

*** See section 7.2.3 'Consenting Issues' and section 6.1.6 'Biota recovery and return techniques'

**** This covers buildings and structures only and does not include spray ponds or cooling canals

Inland river or lake sites

It is less likely that new nuclear stations will be built inland but other large power stations will be sited inland and advice concerning these may be helpful.

Trawsfynydd nuclear plant, now undergoing decommissioning, is the only example of a lake-cooled plant in Britain and it is unlikely that thermal power plants will be built on lakes in the future.

Freshwater reaches of rivers are now considered unsuitable for direct-cooled stations and those, mainly coal-fired stations, constructed prior to the 1960s on large rivers such as the Trent have been phased out in favour of tower-cooled plant. All future river-cooled stations will therefore use tower cooling and where river cooling is not feasible, the option of dry-tower cooling remains open, subject to acceptable energy-efficiency.

The choice of which cooling system to use will depend mainly on planning issues. Natural draught towers are thermally efficient but are large and visually intrusive, particularly when creating vapour plumes. They rarely cause problems with local fogging or ground icing that may be found with lower profile mechanical draught towers. There have been no instances of *Legionella* outbreaks or other health problems associated with either type. Dry-cooling systems have none of these problems but are the least thermally efficient and therefore score badly on relative carbon emissions.

Estuarine sites

Direct cooling is still the most common method on Britain's estuaries, as large tidal fluxes provide efficient heat dispersal. However, estuarine locations are more sensitive than open coastal sites for three reasons. First, they represent important nursery grounds for many fish species such as sole and bass, and any impacts will have implications for the whole stock; secondly, they act as migration corridors for diadromous fish species, including salmon, sea trout, eel, shads, lampreys and smelt, as well as the young of various other fish species. Finally, estuaries are more sensitive to summer heat build-up, as flushing rates are lower than on the open coast and also large intertidal areas absorb solar radiation.

Mixed cooling is an option for estuarine sites. Barking power station on the tidal Thames runs for much of the year on direct cooling but is required under its consents to switch to tower cooling during the summer months when the river temperature exceeds 21.5^{o}C. The Doel nuclear plant, sited on the Scheldt estuary in Belgium, operates under a similar arrangement. Options include a full switch to indirect tower cooling at critical times, or bringing helper towers into circuits (Figure 4-15).

Two further factors affect the future viability of estuaries for direct cooling, particularly in the south of Britain. Background sea temperatures have risen over the past few decades and are expected under various climate change scenarios to continue to rise. UKCIP (2002) predictions suggest possible temperature rises in the southern North Sea and Eastern English Channel of between +2.5 and 4^{o}C (low to high greenhouse gas emissions scenarios) by 2080. Continuing background temperature rises are likely to restrict the ability of cold-water species such as salmon, sea trout and smelt to occupy southern estuaries. The second factor concerns organic (mainly sewage) pollution, which causes summer oxygen sag in many of our larger urbanised estuaries. It is usually a combination of low DO and high summer temperature that limits cold-water fish species (Turnpenny *et al.* 2006). Improvements in sewage treatment works to meet new WFD DO standards is expected to increase summer DO levels, which will have the effect of providing more thermal headroom for cold-water species; where this occurs, it will extend the future of cold-water species in southern estuaries.

Hard-and-fast rules about the type of cooling system to be used on estuaries are best avoided. Each case should be examined according to background temperatures, size of estuary, hydrographic conditions and environmental sensitivities.

Coastal sites

Tower cooling has not been used on any UK or European coastal site to date, direct cooling being the rule. Coastal sites do not suffer the same thermal capacity limits as estuaries, and provided that outfalls are carefully sited to avoid sensitive habitats heat disposal is not a problem. Given the better thermal efficiency of direct cooling, this would be the preferred option, provided that impacts associated with abstraction are acceptable.

7.2.2 Intake structure design and position

All intake types

The type of intake design will depend mainly on its position, which may be onshore, for example as an opening in a wharf (Marchwood, Kingsnorth), onshore via dredged channel (Fawley, Uskmouth, Oldbury), or offshore (most UK nuclear stations).

The primary consideration in selecting an intake position should be to avoid thermal recirculation. This would represent a major environmental impact through loss of thermal efficiency and therefore increased carbon emissions to replace lost efficiency. Other key concerns are to design an intake that is safe to humans and wildlife.

No formal human safety guidelines exist for intakes but the main requirements are that intakes should:

- not create a surface vortex that might endanger craft or swimmers;
- have entrances protected by bars to prevent entry;
- maintain velocities across the face of the bars that would not pin a swimmer or diver to the bars.

These criteria would also protect aquatic mammals. Bar spacing and velocity criteria selected for fish exclusion should be more than adequate for human safety.

Most important for protecting wildlife is to avoid proximity to important fish spawning and nursery grounds. How close is too close? The studies mentioned at Bradwell and Pembroke (Section 6.1.6) provide some insight, where in both cases herring spawning grounds were 10-12 km from the intake positions. In the case of Pembroke, field surveys established that entrainment was unlikely to exceed 0.1 per cent of the Milford Haven adult stock. The Bradwell study showed that up to a quarter of the available herring larvae and post-larvae might wash past the intake, but it was not established how many of these would enter the intake, although concentrations in the cooling water appeared to be only around 10 per cent of those in adjacent open water (Coughlan *et al.* 1980). However, it would be dangerous to apply any simple distance rule-of-thumb as the risk to stocks depends greatly on the local hydrographic conditions and where the intake lies relative to the drift path from the spawning ground. A single tidal excursion might carry spawn or larvae over 10 km or so. It is necessary, therefore, to undertake a hydrographic analysis similar to those described in Section 6.1.6, or to measure actual concentrations of ichthyoplankton in the abstraction field of the intake.

Different principles apply when considering proximity to nursery grounds, as by the time fish have graduated to nursery areas they are no longer planktonic, therefore less at the mercy of currents. More effective mitigation measures can be provided in the form of fish deflection and return systems. Less reliance will be placed on mitigation measures when the intake is located in an area naturally low in juvenile fish. This means avoiding as far as possible abstraction from intertidal areas, including estuarine foreshore, saltmarshes and the surf zone of beaches, except where it can be shown that the habitat is relatively unproductive. Further specific advice is given below.

Onshore intakes

Positioning

Onshore intakes that use dredged channels run the risk of entrapping fish from highly productive intertidal habitat, the fish tending to run into deeper channels towards low tide (Figure 6-12). In estuaries, intakes of this type are also likely to intercept juvenile fish that migrate along the channel margins. One option might be to sheet pile the channel sides so that intertidal drainage runs away from, rather than towards, the channel. Dredged onshore intake channels across saltmarshes or other high quality habitats are best avoided entirely.

Wharf-type onshore intakes that open directly into deep water are less of a risk to intertidal biota and are the preferred option where an onshore intake is used.

In either case, it is important to avoid locating the intake in a backwater area where large quantities of weed or other debris might accumulate. Where stations are to be built on existing power station sites, historical knowledge of screen blockage problems should be reviewed.

Design of structure

The main requirement is to ensure that intake openings are large enough to keep approach velocities below recommended values. Environment Agency Best Practice (Turnpenny and O'Keeffe, 2005) allows a flexible approach to setting intake velocities, using data on fish swimming speeds. This ensures that the intake structure is no larger than necessary, keeping down costs, the physical and carbon footprint and reducing sedimentation risk associated with low water velocities. In practice, entrance velocities at maximum abstraction rate normally fall between 0.3 and 0.5 ms^{-1}.

Fish deflection measures

AFD systems, strobe lights or other deflection devices are normally installed in or immediately in front of the intake openings. In rarer cases, they are mounted on specially constructed frames or piles placed some distance upstream of the structure where water velocities are lower (see Figure 6-19). For new plants it is preferable to design the intake openings to have suitable velocities in the first place. For AFD systems, acoustic modelling is an essential step in the design process to achieve an effective but confined sound field.

Offshore intakes

Positioning

Offshore intakes take abstractions away from sensitive intertidal nursery areas but the precautions listed above for spawning areas still apply. Locating an intake offshore may affect commercial fishers by interfering with trawl lines. It may in some cases increase the risk of taking in species migrating along the coast, such as salmon (for example, on the north-east coast of England, where T-net interception fisheries operate) or of sprat inundations (predominantly North Sea and Eastern English Channel).

Design of structure

Offshore intake structures used in the past have not usually been designed with fish protection in mind, or at least not with a good understanding of the subject. There is no excuse nowadays for building a structure that suffers from the problems highlighted in earlier sections of this report (6.1.6). The two main offences are (1) allowing tidal stream velocities to add to inlet velocities, and (2) designing a capped structure that does not conform to the hydraulic requirements for a velocity cap. The LVSE intake (Figure 6-17) provides one possible design but has not yet been developed and tested for nuclear power stations. Any design that can be shown in model tests (preferably with physical modelling) to meet the velocity requirements should be acceptable.

Biota deflection measures

As for onshore intakes.

7.2.3 CW screening and fish protection systems

Choice of screen type

Primary screening at the intake point using passive wedge-wire cylinder (PWWC) screens, although in theory scalable to handle tens of cumecs CW flow, is unlikely to be acceptable to nuclear operators owing to unproven viability in the hostile offshore conditions of the UK, blockage risk, and difficulties of access for maintenance.

Fine screening using band or drum screens can provide suitable protection for the main cooling circuit as well as for entrapped biota (using FRR techniques – see below). The preferred mesh size may depend on the developer, European stations traditionally using finer (2-3 mm) screens than British counterparts (6-10 mm). For FRR purposes, a screen size of up to six mm is advised but meshes this small or smaller at coastal sites may lead to a risk of ctenophore blockage during the summer months.

Use of intermediate raked bar screens may be required at some sites. Their use is not desirable from a fish protection point of view but, if used, consideration should be given to safe and efficient fish handling.

Fish deflection/fish recovery and return systems

Detailed advice on these subjects is given in the Environment Agency Best Practice Guide (Turnpenny and O'Keeffe, 2005) and only outline details are given here.

Where PWWC screens are not used, best practice for directly cooled coastal sites allows a combination of acoustic fish deterrents (AFDs) and fish recovery and return (FRR). The combination of both processes caters for hearing-sensitive, delicate species (AFD) as well as more insensitive demersal and epibenthic species, including crustaceans (FRR).

Although the best practice guidance is relatively recent, fish protection technology is moving fast, partly in attempt to keep up with developing European legislation (notably the eel management regulation and new fish pass and screens regulations proposed for the 2009 parliamentary session to bring controls up to Water Framework Directive standards. Consequently, it may be necessary to go beyond current best practice, with better provision to protect eels. This will require the development and trialling of larger fish buckets for band and drum screens, more suited to handling sinuous species such as eel and lamprey. Fish deterrent strobe lights also appear promising for eel deflection. Accordingly, the recently granted Abstraction Licence (Ref. No. 22/61/6/156)

for Pembroke CCGT station includes a requirement to install and test the efficacy of strobe lights at the intake.

Some problems still need to be resolved with FRR systems. In the case of band screens, if larger fish buckets are to be used, openings through which the band screens pass may need to be enlarged to allow the larger buckets to pass through. For new systems built from scratch this should not be a problem. Where older facilities are being reused, as has been the practice at numerous recent new builds, openings may require modification. On standard UK drum screens, fish buckets are fitted inside the drum and this is not an issue. Hopper geometry on drum screens does, however, need to be checked (see Section 6.1.6).

Some difficulties have arisen in the application of best practice for FRR systems. One aspect concerns the requirement not to chlorinate ahead of the screenwells, to avoid exposing fish to the toxic effects of the biocide. While concentrations and exposure times may not be directly lethal, to use chlorine before this point is likely to disorientate fish and leave them more vulnerable to predation at the return point. Where there is a risk of biofouling upstream of the screens becoming an operational threat, plant designers/operators need to find effective solutions. These could for instance include periodic shock dosing of individual screenwells when offline, treatment of surfaces with fouling-resistant paints or gels or manual cleaning as required.

The advice that bends in launders from FRR systems should have a minimum radius of three metres has also been difficult to achieve in some cases, especially where old infrastructure has been reused with insufficient space to achieve this. The purpose of this criterion is to ensure that material does not become caught up in bends, restricting water flow and trapping fish. Where this cannot be achieved, the developer should be able to demonstrate that this will not happen, for example by using larger channel sections and ensuring adequate water flow.

The predation rate at FRR outfalls has received little attention to date, even though experience in similar situations suggests that predators are attracted to any discharge containing potential food items. This aspect would merit further consideration.

In-plant monitoring facilities

Abstraction licence conditions for recently consented power stations have included a requirement to monitor impingement and entrainment rates. Specifying sampling points at the design stage facilitates this.

Entrainment sampling

Sampling on older plants has normally required dropping plankton pump samplers or plankton nets into the CW forebay. The procedure is unwieldy and time-consuming and, where tethered nets are used, turbulence makes quantification difficult.

A better option is to draw water from the CW culverts via a tapping on the pressure side of one of the CW pumps. At this point, any young fish will be in the dark and unlikely to orientate to avoid entrainment into the sampling offtake. The sampling offtake should use at least a 100-mm inside diameter tapping and be capable of delivering at least 25 l s^{-1} flow. It may be worth installing, say, three tappings side by side so that sampling can be done in triplicate.

A standard pipe tapping would draw water primarily from the boundary layer of the CW culvert, which may not be ideal. The alternative would require the sampling point to

project at least some distance from the culvert wall. No design was found for such an arrangement but a simple hydraulic analysis, for example using CFD modelling, would enable the development of a suitable design.

In some cases entrainment sampling points have been taken from screen washwater pump supplies. This arrangement is second best as the much smaller size of the pumps will mean that a large proportion of the ichthyoplankton is likely to become damaged, making identification difficult.

The sample is collected by filtering the sample flow through a plankton net suspended in water, normally for a 24-hour period (though maybe split into shorter subsampling periods). The facility must therefore incorporate some sort of tank and a drain. A suitable arrangement is shown in Figure 7-1.

Impingement sampling

The arrangement in Figure 7-1 also caters for impingement sampling. Material normally returned to source via the FRR launder is diverted into a basket in a collection well. Water level in the well is maintained by a weir. The set-up is similar to many trash basket arrangements at power stations, except that a weir is used to maintain a water level. On new nuclear stations, owing to CW requirements, the amount of material passing through the system will be large at times and sampling systems must be able to cope with this. If there are four screens it may be necessary to have four separate sampling wells, or one shared between two screens. Lifting gantries will be required.

To reduce the risk of fish exhaustion or damage during retention in the collection well, the design should ensure that energy dissipation within the well is kept at or below 100 Wm^{-3} (see 6.1.6).

Consenting issues

Discharge of FRR systems to sea

There has been some confusion in the past about the status of backwashings from CW screens and whether their discharge to sea should be treated as a trade waste. The precedent of returning backwashings to sea was established for Sizewell B, owing to the perceived ecological benefits of returning both live and dead material to the ecosystem (Turnpenny and Taylor, 2000). A number of new stations have followed suit. The Environment Agency has no formal policy, but the following may be helpful:

- If the waste is returned to the water in a continuous stream, along with entrained material, there should be no consenting issues. This reduces landfill waste.
- Maceration of screen arising may be acceptable if discharged continuously. A degree of maceration occurs incidentally on some current, more primitive fish return systems.
- Concentrated dumping of accumulated waste would be a different matter and could have significant waste and water quality issues.

On the last point, consenting of the Sizewell B fish return system required a means of monitoring fish throughput in the return system (visual observation/CCTV camera) and that the fish return launders should be diverted into trash baskets if abnormally high numbers of fish were being returned to sea. The provision was made owing to previous experience at Sizewell A of sprat inundations, and concerns that large numbers of dead fish might wash up on local beaches. As part of the commissioning phase studies at Sizewell B, over a period of six weeks releases of thousands of dye-marked dead

sprats into the CW discharge were made. During this period, National Rivers Authority staff monitored Sizewell Beach but recorded no incidence of marked sprats being found on the beach, and no higher incidence of dead fish than when the plant had not been operating (Turnpenny and Taylor, 2000).

All of the above indicates a need for case-by-case consideration. The issue of reducing fish kill will influence future policy and practice over biodegradable waste of this origin.

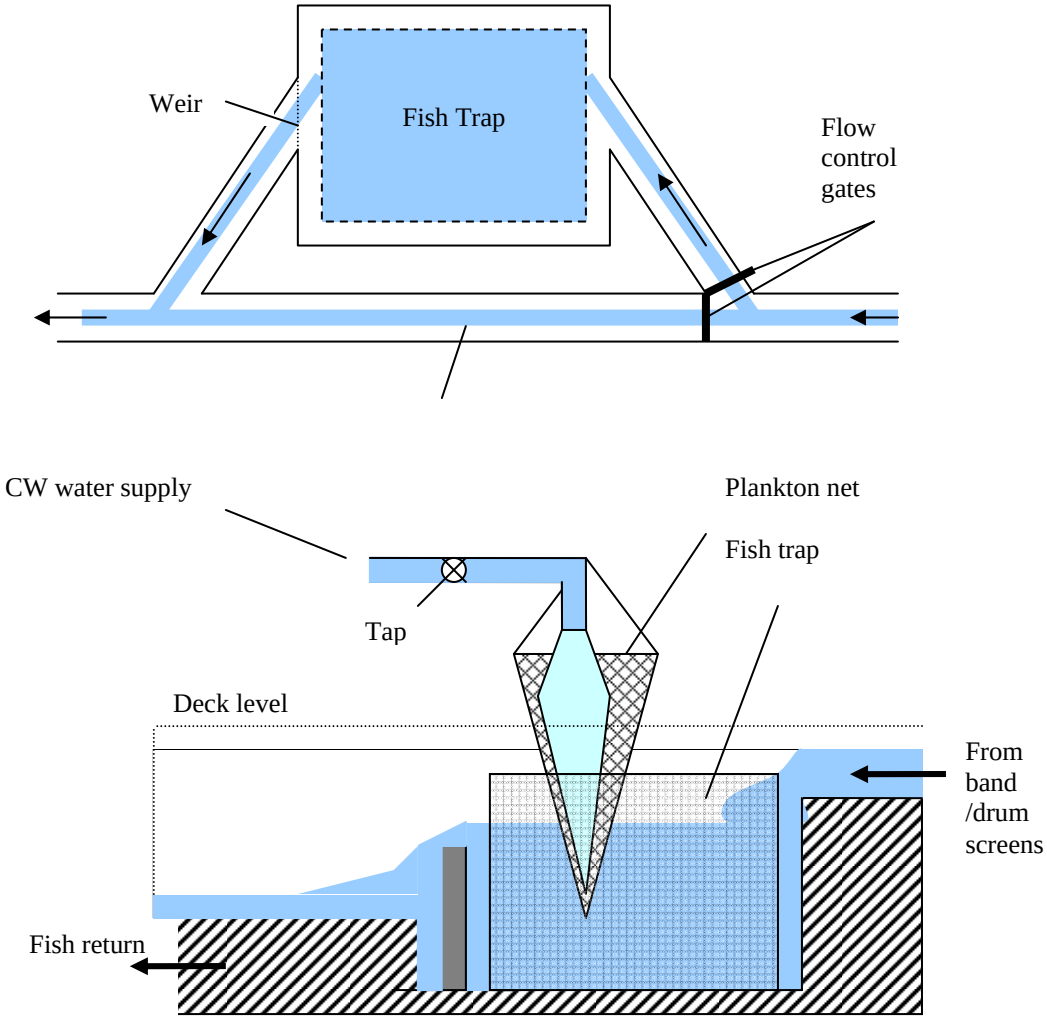


Figure 7-1 Example of sampling facilities for impingement and entrainment monitoring. Upper drawing shows diversion from FRR launder into water-retaining collection basket for impingement sampling. During sampling, shutters or gates are adjusted to divert flow into sampling well. Lower drawing shows the arrangement in vertical section, along with the suspended plankton net used for entrainment sampling.

Trialling mitigation techniques

Useful development of improved biota protection techniques can often be achieved in laboratory flumes or other systems but trialling necessarily involves testing on live plant. Fish deflection systems are normally tested by comparing impingement rates over alternating on-off periods. Excessive regulation can prove a hindrance to this process. For example, the introduction of combined AFD and FRR techniques at

Shoreham CCGT power station in the 1990s led to the lowest recorded rate of fish impingement for any UK station investigated up to that time (Clough *et al.* 2003) but the true efficiency of the AFD system could not be investigated as the abstraction licence required the mitigation measures to run at all times when the plant was operating. In more recent abstraction licence negotiations, it has been agreed that this requirement can be relaxed for the purpose of scientific testing of mitigation measures. This approach could usefully be adopted at other new build sites.

7.2.4 Cooling towers

It is unlikely that nuclear station developers would opt for indirect cooling by choice, owing to the various penalties: planning issues, land-take, capital and running costs, lower efficiencies, waste-disposal and so on. If forced by aquatic environmental issues to use indirect cooling, tall natural draught towers in the coastal landscape are unlikely to be acceptable to planners for aesthetic reasons, whilst dry cooling or condensing would be an unlikely choice on cost and efficiency grounds and is only defined as BAT where water supply is limiting. This leaves large arrays ("streets") of mechanical draught towers. Since the most likely nuclear sites are coastal or estuarine, the towers would be using seawater and so in addition to planning, cost and efficiency, there would be issues of drift, carryover, circuit concentration factors and coping with the often high suspended sediment loads in estuaries.

7.2.5 CW circuit

As ichthyoplankton entrainment represents one of the most difficult impacts to mitigate, attention must focus on aspects that are amenable to control:

- Initial site selection and intake positioning to minimise concentrations of ichthyoplankton in the abstraction zone.
- Design and operational aspects of the cooling system that influence survivorship of cooling system passage.

The first of these was dealt with earlier in this section. With regard to cooling system design, it is too early to make recommendations on pressure profiles within systems, as data are relatively sparse; at first sight, this would seem an unlikely route, as these conditions are largely determined by physical dimensions of the circuit which can probably not be changed much. Equally, temperature and exposure time probably cannot be altered. This leaves biocide toxicity as the main factor that might be altered. In some locations, where biofouling risk is low, it may be possible to avoid biocide application altogether (as is presently the case at Hinkley Point). Where biocide is used, it may be possible to modify dosing regimes to minimise entrainment mortality. This would require good knowledge of dose-response curves under simulated passage conditions (see Section 6.1.4). Where chlorine or other biocides are used, they are normally applied only for the active growth season, when water temperature exceeds 10^{o}C (roughly May-October), or as shown to be required by monitoring biological growth.

7.2.6 Outfall design and position

Optimising outfall location requires first the identification and mapping of sensitive habitats and biotopes, along with sensitivity criteria (such as maximum allowable exposure temperature or maximum allowable daily temperature change). Sources of

data are identified in Section 6.3. Inspection of sensitivity maps will usually suggest a suitable location, which can then be investigated further using one of the plume or long-term heat field models listed in Section 6.3.4. Such analyses also seek to minimise recirculation risk between intake and outfall and may require several iterations.

The behaviour of the plume and areas swept by temperatures above +3^{o}C ΔT (current maximum temperature uplift outside the plume in UKTAG draft standards) should be investigated by modelling; no sensitive habitat or biotope should be exposed to the plume for sufficiently long, or often or at temperature/ concentration sufficient to cause harm. Plume attachment to the shore should be avoided. The draft standards do not specify the allowable spread of the plume across an estuarine channel, whereas SEPA (2006) guidance states that the mixing zone for any type of discharge into transitional water should not take up more than half of the narrowest dimension. A more widely used criterion in overseas regulation that was also proposed by the authors of the technical report supporting the development of UKTAG draft standards (Turnpenny and Liney, 2006) requires the plume to occupy no more than 25 per cent of the channel cross-section for no more than five per cent of the time (allowing for some spread at slack water). It was excluded from the draft standards owing to potential enforcement difficulties, but this criterion provides clear guidance when interpreting plume model outputs and its adoption for this purpose is recommended.

Achieving an acceptable thermal field may require not only testing of alternative outfall positions but introducing variations in outfall design, for example by using diffusers.

7.2.7 Effects of long-term processes on cooling system viability

Climate change

This report does not cover climate change predictions or impacts but some indication is needed, in general terms, on what rising temperatures might mean for the operation of power plant cooling systems. Should background water temperature become one degree warmer than it is today then, by and large, water coming out of a direct-cooled station also will be one degree warmer. This is what currently happens but with a warming of 15ºC or so winter to summer, and the cooling system is designed to operate across this broad range of environmental conditions. As outlined in Chapter 2, maximum thermal efficiencies for steam turbines are achieved with low backpressure, typically 20 to 50 mbarg (2 to 5 kPa) equivalent to saturation values (condensation temperatures) of 18 to 33ºC. Efficient and economic transfer of heat from the tube surface to the cooling water requires a temperature differential ΔT of about 10ºC, so the cooling water source should not exceed 23ºC if 50 mbarg is to be maintained.

Table 7-3 Turbine backpressure and steam condensation temperature (saturation value) for a range of nominal CW intake temperatures (10ºC ΔT)

Backpressure mbar gauge	Condensation temperature ºC	Nominal intake or recool temperature ºC
20	17.5	8
30	24.1	14
40	29.0	19
50	32.9	23
60	36.2	26
70	39.0	29
80	41.5	32
84 Damhead Creek (design)	43.8	Air-cooled
150 Damhead Creek (extreme)	52.0	
186 Matimba S. Africa	59.0	

Direct-cooled CW systems can be designed for worst-case operation, like the highest recorded local sea temperature plus the maximum estimated re-circulation, plus a bit extra. Accommodating these marginal temperatures incurs additional cost and could result in excess cooling capacity for most of the station's life, particularly in winter when even conservatively designed stations can shut down one CW pump, thereby decreasing the rate of flow through the condensers with an increase in ΔT, assuming that the discharge consent allows this. Thus, a station can be designed in one of two ways: a low rate of abstraction and high ΔT, or a high rate of abstraction and low ΔT. Each approach has environmental advantages and disadvantages. It is usual, particularly today, to design around a reference temperature at the CW intake that will give adequate cooling for a percentage of the year whilst accepting some loss of efficiency (higher back pressure) for a few days, or even weeks, each summer. Electricity demand and prices are lower in summer anyway - although this could change with more air-conditioners being installed.

Current warming predictions appear to be around 0.1 to 0.2ºC per decade so, given a station lifetime of around six decades, it is unlikely that major changes to current design philosophy will be required. Towards the end of its working life, when predicted temperature rises might reach 1.0 ºC, the station might no longer be required to reach maximum output - assuming that the owner also operates newer stations. Efficiency per se has not been an issue with nuclear plant as it is for conventional plant since fuel represents a far lower proportion of generation costs. Nuclear plant is relatively inflexible in operation and historically has provided the baseload. If nuclear comes to represent a large proportion of UK installed capacity or if there is a major increase in intermittent renewables, such as wind, with privileged access to the grid, older nuclear plants could be forced into intermittent operation anyway.

The warm water discharge does not necessarily take longer to cool in summer, nor to spread more widely. Summer air temperatures are higher, as is the water/air saturation value, so evaporative cooling is enhanced. Consequently the atmospheric exchange coefficient AEC (net loss of heat from water to air) rises. On the other hand wind speeds tend to be lower in summer and this reduces cooling; some climate change models predict that these will increase.

There is little scope for amelioration. Station designers look for the coldest, usually the deepest, water source but the length of intake tunnel that can be built is constrained by

economics. Recirculation of discharged water at some states of tide, or a slight build-up of heat in the area, might also be too expensive to avoid completely. The size of condensers, and hence abstraction rates, could be increased, but this also has monetary and environmental costs that may not be sustainable.

For river (tower-cooled) stations higher air temperature will enhance evaporative cooling, but at the cost of increased evaporative water loss and increased make-up requirement. The air/water ratio in the towers could be increased to obtain better recool but this would mean larger towers and more evaporative loss. High air temperature reduces the efficiency of dry towers and air-cooled condensers.

One of the many predicted effects of climate change is a modification to the seasonal pattern of rainfall and increased risk of water shortages in southern and eastern Britain. This could be a further reason to avoid wet tower-cooled freshwater sites.

Water levels can be expected to change as a result of climate change. Relative sea level rise is expected to be between 0.17 and 0.77 m by 2080 (UKCIP, 2006). Effects on cooling systems per se should be minor, or beneficial where dredged intake channels are used, in terms of increasing available water depth.

Geomorphology

Within the long life-cycle of a nuclear station, coastal processes in some dynamic locales may lead to major shifts in bathymetry that could threaten CW supply and alter patterns of heat dispersion. Moderate sedimentation can be dealt with by dredging but movement of offshore banks to cut off flow around the intake and outfall zones could reduce coastal water exchange and limit cooling capacity. It is therefore important to consider coastal processes at the planning stage for large plants.

7.3 Status of BAT definitions for cooling water systems

The European Commission Reference Document on Application of Best Available Techniques to Industrial Cooling Systems (BREF Cooling, December 2001) identifies direct cooling as BAT for large power plant cooling systems, stated as follows:

"*In an integrated approach to cooling an industrial process, both the direct and indirect use of energy are taken into account. In terms of the overall energy efficiency of an installation, the use of a once-through system is BAT, in particular for processes requiring large cooling capacities (e.g. > 10 MWth$_h$). In the case of rivers and/or estuaries once-through can be acceptable if also:*

- *extension of heat plume in the surface water leaves passage for fish migration;*
- *cooling water intake is designed aiming at reduced fish entrainment;*
- *heat load does not interfere with other users of receiving surface water.*

For power stations, if once-through is not possible, natural draught wet cooling towers are more energy-efficient than other cooling configurations, but application can be restricted because of the visual impact of their overall height."

It may be concluded from the BREF that direct cooling would not be BAT for large power stations if any of the three conditions were not met.

With regard to reduction of “entrainment” (for this read “entrainment and impingement” or “entrapment”), BAT requires positioning of the intake to avoid sensitive biotopes, spawning and nursery areas and fish migration routes. The BREF goes on to say “*From the applied or tested fish protection or repulsive technologies, no particular techniques can yet be identified as BAT*.”

The validity of direct cooling as BAT has been challenged by consultants acting on behalf of Countryside Council for Wales (Cambrensis, 2008). Their report concludes: *“that both the developer and EAW[35] should fully consider alternatives to direct cooling for this installation at this site, particularly give significant developments in BAT since the European BREF guidance was last assessed in 2001, particularly in the USA, where they no longer regard direct cooling as best available technologies for coastal power stations”.*

The Cambrensis conclusion has implications in a broader UK context and needs to be considered for future cooling requirements of large-scale power developments. Present indications are that direct cooling would be the method of choice at most, if not all, new UK nuclear developments.

The first point to consider is the relative thermal efficiency of available cooling methods, which has implications for the cost-per MWh of running the plant and levels of carbon emissions, assuming that any nuclear-generated electricity would displace the equivalent fossil-fuelled generation. The relative efficiencies remain as stated in Section 7.1.5 above, with direct cooling being the most efficient.

Secondly, if direct cooling is BAT on energy efficiency grounds, it needs also to meet the requirements of BREF listed above. The first of these deals with fish migration and would be met using advice identified in Section 7.2.6 above, which takes account of the latest understanding of temperature effects on fish migration developed as part of new WFD water temperature standards (UKTAG, 2008); this is augmented by new data on effects of temperature steps on juvenile fish migration, presented in Section 6.3.7.

Regarding impingement and entrainment of biota, the Cambrensis report notes advances in cooling technology since 1997 (the latest reference cited in BREF) but does not acknowledge progress made in entrapment mitigation. By 1997, UK power stations were operating only primitive fish return systems and deflection techniques for large water intakes had not advanced beyond the use of bubble curtains (Heysham I and II). Development of acoustic fish deterrent technology was in its infancy. In 2005, the Environment Agency and conservation bodies published their best practice guide on intake and outfall fish screening (Turnpenny and O’Keeffe, 2005), which identifies a range of techniques and combinations of AFD and FRR suitable for directly cooled coastal and estuarine plant. Now, a number of coastal power stations, along with numerous other water abstractions, use these techniques and the technology can be considered mature. Thus, an updated BREF might be expected to draw more positive conclusions on ‘fish protection and repulsive technologies’.

The Cambrensis conclusion also refers to the situation in the USA, where there is a presumption against direct cooling (though not a ban) for large power plants. A distinct difference in the US approach has been the assumption of 100 per cent mortality of any fish eggs, larvae or juveniles entrained in plant cooling systems and discharged back to sea. UK studies have shown that substantial proportions survive CW system passage, potentially reducing the magnitude of entrainment impacts.

We therefore conclude that direct cooling may be the best environmental option for large power stations sited on the coast or estuaries, subject to current best planning, design and operational practice and mitigation methods being put in place, and meeting conservation objectives of the site in question.

[35] Environment Agency wales

7.4 Dealing with residual impacts

For a major project to proceed, impacts remaining after all mitigations have been applied must be deemed acceptable or compensated in some form. Recent years have seen much progress in the development of ecological compensation measures, both overseas and in the UK. The main requirements are that compensation should as far as possible be like-for-like and commensurate with, or an improvement upon, the level of impact. In practice this means exceeding the estimated loss, as replacement measures are often of lower quality than the original or may take time to develop.

Identifying suitable compensations sometimes requires imagination. Some years ago, a compensation plan was developed for a directly cooled plant proposal on the Usk Estuary. The environmental statement identified potential losses of salmon smolts to impingement and issues relating to upstream migration past the thermal plume. Compensation measures considered included a buy-out of the Usk salmon net fishery, improvements to fish passes and improvement of riparian habitat on the inland R. Usk. More recently, compensation habitat for UK port developments has been provided through managed coastal realignment projects.



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Bird Study (1985) **32**, 199–206

The food of breeding Marsh Harriers *Circus aeruginosus* in East Anglia

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In East Anglia in 1983–84, Marsh Harriers preyed upon most species of birds and mammals present in their hunting areas. Small birds, young gamebirds and waterbirds and mammals formed 90–95% of the diet by numbers and weight. Small adult birds and mammals were important in April–May, but thereafter young gamebirds and waterbirds formed an increasing proportion of the diet, whilst the proportion of mammals declined.

Females and polygynous males took larger prey after the young had hatched than did monogynous males. There was no difference in the proportions of large prey taken by polygynous males and the much larger females. There was some evidence of a difference in diet for monogynous males, in the latter part of the season, when they took fewer mammals and more small birds. Polygynous males took larger prey later in the season, mainly by taking larger individuals of the same species. There was no evidence that the proportion of large prey brought by males to individual nests was linked to brood size.

No previous quantitive data have been published on the food of Marsh Harriers in Britain. The most recent available data were from Holland and France during 1964–71 (Schipper 1973). They showed that a wide variety of prey was taken, with young Pheasants and waders in Holland and northern France, Rabbits in Holland, and snakes and young waterbirds in southern France being particularly important. Schipper identified 852 items, including 167 from nest hides and the remainder by methods similar to those described here. This study is based on 1007 identified items and 1067 items classed only by size.

The study area comprised some 50 km^2 of intensively farmed arable land intersected by grass-covered banks, farm roads and ditches. There were small areas of woodland, a few hedges, and some old pasture, but most of the ground was cultivated for cereals, sugar beet, potatoes, carrots, peas, beans and oilseed rape. Gamebirds were not artificially reared in the area, although well-grown Pheasant poults were released.

The Harriers nested mostly in small reedbeds but with some nests in arable crops. In 1983, there were three pairs plus a polygynous male with three nests. The following year three pairs again nested, plus two bigynous males with two nests each. All 13 nests were successful. In 1984 the male of a pair disappeared two days before the first egg hatched, and the female's prey was artificially supplemented by placing rabbits and ducklings near the nest site. Care was taken to exclude these supplemented items from subsequent analysis of prey remains and pellets.

METHODS

Pellets and prey remains were collected on the limited number of occasions when nests were visited for other purposes, and from the nest area soon after the young had fledged. Regular searches were also made of farm tracks, edges of reedbeds and ditch-sides that the birds were known to use regularly as resting and feeding places. In addition, some items were identified

Table 1. Marsh Harrier prey identified from pellets, prey remains and sightings 1983–84

	Pellets				*Prey remains*				*Sightings*			
	Nos	%	*Wt*	%	*Nos*	%	*Wt*	%	*Nos*	%	*Wt*	%
Reed Bunting *Emberiza schoeniclus*	8	2.0	146	0.3	3	1.0	55	0.1	2	0.6	37	0.1
Meadow Pipit *Anthus pratensis*	17	4.3	340	0.6	6	2.0	120	0.3	7	2.2	140	0.3
House Sparrow *Passer domesticus*	11	2.8	297	0.5	4	1.4	108	0.2	2	0.6	54	0.1
Unidentified small birds	45	11.5	1125	2.2					36	11.2	900	2.1
Greenfinch *Carduelis chloris*	1	0.3	28	0.1	4	1.4	111	0.2				
Skylark *Alauda arvensis*	12	3.1	456	0.9	8	2.8	304	0.6	7	2.2	266	0.6
Starling juvenile *Sturnus vulgaris*	11	2.8	770	1.5	39	13.2	2730	5.9	4	1.3	280	0.6
Starling adult *Sturnus vulgaris*	34	8.7	2788	5.3	13	4.4	1066	2.3	5	1.5	410	0.9
Shelduck pullus *Tadorna tadorna*	6	1.5	282	0.5	13	4.4	611	1.3	6	1.9	282	0.7
Partridge pullus *Alectoris rufa-Perdix perdix*									4	1.3	204	0.5
Lapwing pullus *Vanellus vanellus*									4	1.3	340	0.8
Moorhen pullus *Gallinula chloropus*									4	1.3	380	0.9
Pheasant pullus *Phasianus colchicus*									42	13.1	4452	10.3
Cuckoo *Cuculus canorus*	2	0.5	228	0.4	3	1.0	342	0.7				
Mallard pullus *Anas platyrhynchos*					13	4.4	1625	3.5	25	7.8	3125	7.2
Tufted Duck pullus *Aythya fuligula*					2	0.7	250	0.5				
Turtle Dove *Streptopelia turtur*	2	0.5	290	0.5	7	2.4	1015	2.2	1	0.3	145	0.3
Red-legged Partridge juvenile *Alectoris rufa*									6	1.9	1008	2.3
Pheasant pullus/juvenile *Phasianus colchicus*	76	19.4	15,428	29.6	58	19.6	11,774	25.2				
Oystercatcher pullus *Haematopus ostralegus*					1	0.3	205	0.4	2	0.6	410	0.9
Lapwing *Vanellus vanellus*	1	0.3	228	0.4	1	0.3	228	0.5	4	1.3	912	2.1
Duckling sp. pullus/juvenile Anatidae	31	8.0	7378	14.2								
Black-headed gull *Larus ridibundus*					1	0.3	276	0.6				
Moorhen juv/adult *Gallinula chloropus*	6	1.5	1794	3.4	21	7.1	6279	13.4	16	5.0	4784	11.1
Pheasant juvenile *Phasianus colchicus*									33	10.3	9900	22.9
Duckling sp. juvenile Anatidae									2	0.6	740	1.7
Grey Partridge *Perdix perdix*					2	0.7	748	1.6				
Coot juvenile *Fulica atra*	3	0.8	1200	2.3					1	0.3	400	0.9
Stock Dove *Columba oenas*	3	0.8	1200	2.3	8	2.8	3200	6.9				
Red-legged partridge *Alectoris rufa*	2	0.5	968	1.9	3	1.0	1452	3.1	3	0.9	1452	3.4
Oystercatcher *Haematopus ostralegus*									1	0.3	519	1.2
Wood Pigeon *Columba palumbus*	6	1.5	3144	6.0	1	0.3	524	1.1	1	0.3	524	1.2
Pheasant *Phasianus colchicus*									1	0.3	600	1.4
Field Vole *Microtus agrestis*	19	4.9	722	1.4	1	0.3	38	0.1				
Unidentified small mammal									50	15.6	1500	3.5
Water Vole *Arvicola terrestris*	4	1.0	400	0.8	6	2.0	600	1.3				
Rabbit *Oryctolagus cuniculus*	69	17.7	10,143	19.5	58	19.5	8526	18.2	21	6.6	3087	7.1
Weasel *Mustela nivalis*									4	1.3	604	1.4
Common Rat *Rattus norvegicus*	9	2.3	2070	4.0	1	0.3	230	0.5	18	5.6	4140	9.5
Stoat *Mustela erminea*					1	0.3	267	0.6	1	0.3	267	0.6
Brown Hare juvenile *Lepus capensis*	1	0.3	300	0.6	10	3.4	3000	6.4	4	1.3	1200	2.8

Brown hare adult *Lepus capensis*					1	0.3	600	1.3				
Avian egg Phasianidae-Anatidae	6	1.5	252	0.5	1	0.3	42	0.1				
Others	6	1.5	160	0.3	6	2.0	399	0.9	3	0.1	265	0.6
Totals	391				296				320			

Others include:

Pellets: Yellow Wagtail *Motacilla flava* (2); Chaffinch *Fringilla coelebs* (2); Song Thrush *Turdus philomelos* (1); Shrew sp. Soricidae (1).

Prey remains: Sedge Warbler *Acrocephalus schoenobaenus* (1); Linnet *Acanthis cannabina* (1); Tree Sparrow *Passer montanus* (1); Blackbird *Turdus merula* (2); Jay *Garrulus glandarius* (1).

Sightings: Redshank pullus *Tringa totanus* (2); Blackbird *Turdus merula* (1).

from fur, feathers and other remains at places where the birds had recently caught prey.

Sight identifications of prey carried by the adults (sightings) were obtained by watching the birds returning to the nesting area carrying prey, almost invariably in their feet. Several nest sites were watched from a distance and all prey being delivered by the adults, whether identified or not was classed by size. Observers were given a list of prey species categorized as small, medium or large, and asked to class all prey items by reference to comparably sized species on the list. Listed items were chosen to represent the weight classes: small (under 125 g), medium (125–250 g) and large (over 250 g), but few large items were recorded, so these were combined with medium items for analysis. Prey classed but not identified was included in all calculations for sightings.

Food items in pellets and prey remains were identified by reference to a collections of skins and feathers made during the study of mammals and birds which were known to occur in the area during the breeding season. Only one item of each species identified was ascribed to individual pellets, which usually contained the remains of more than one prey item of different species.

Some invertebrate remains were found in pellets but they could have been picked up accidentally or formed part of the diet of prey species; they were excluded from analysis. A few items were weighed in the field but weights of adult prey species were mostly taken from Hickling (1983), Corbet & Southern (1977), and Schipper (1973), and for juveniles estimated from growth curves derived from Baur & Glutz (1966), Pulliainen (1965), and R. Green (pers. comm.). Thus some estimated prey weights, particularly of juvenile birds, were approximate. Records from pellets, prey remains and sightings have been treated separately in the analyses to avoid duplication where prey from sightings subsequently turned up as remains or pellets. The data from the two years were pooled, as no significant differences emerged between the years.

The scientific names of all prey species have been included in Table 1.

RESULTS

Overall diet

All the identified prey items for 1983 and 1984 are shown in Table 1 with the exception of invertebrates (39 Coleoptera, one Lepidoptera, one Mollusca). The most numerous items identified from pellets and remains were young Pheasants, Rabbits and Starlings, with

Table 2. Percentage occurrence of the main prey categories in the diet of Marsh Harriers by numbers (*N*) and by weight (wt)

	Pellets		*Prey remains*		*Sightings*	
	N	*Wt*	*N*	*Wt*	*N*	*Wt*
Small birds	35.8	11.7	27.1	10.5	19.9	4.7
Gamebirds	19.9	31.5	19.6	25.2	26.9	37.4
Waterbirds	12.9	20.8	18.9	24.3	21.3	31.4
Mammals	21.3	26.3	26.2	28.5	28.5	24.9
Others	10.1	9.7	8.2	11.5	3.4	1.6

young Pheasants, small mammals and Mallard ducklings in sightings.

On the basis of biomass, young Pheasants again contributed most in all three samples, with Rabbits second for both pellets and prey remains and Moorhens in sightings. Overall, small birds, gamebirds, waterbirds and mammals formed 90–95% of the diet both by numbers and weight in all three samples (Table 2).

Live weights varied between 10 and 600 g, although both items at 600 g were estimated. These were a female Pheasant from a male Harrier, and sub-adult Hare from the female which lacked a mate. Female Harriers are some 30% heavier than males (means 669 and 500 g respectively, Cramp & Simmons 1977). A male was seen carrying an adult Wood Pigeon (450 g), killing an adult Red-legged Partridge (500 g) and a Leveret (300 g). All these items were recovered complete and weighed.

Mean live weight per item from pellets was 133 g, from prey remains 158 g and from sightings 135 g; 20–24% of female and 27–32% of male Harrier body weight respectively.

Table 3. Size of prey delivered by monogynous males, polygynous males, and female Marsh Harriers

	Total items delivered	*Percentage under 125 g*
Monogynous males (5 birds)		
Courtship	22	81.8
Incubation	84	82.1
Nestling	227	83.3
Post-fledging	7	100.0
Polygynous males (3 birds)		
Courtship	7	57.1
Incubation	187	83.4
Nestling	314	69.7
Post-fledging	22	63.6
Females (9 birds)		
Nestling	182	74.7
post-fledging	15	80.0

Seasonal changes in diet

Prey size

All prey seen delivered to nesting areas was classed as small or large. Eighty-three per cent of all items thus classed were small (under 125 g). Table 3 shows the percentage of small items for monogynous and polygynous males for four phases of the breeding season, and for females for two phases (no prey was seen to be carried by females prior to hatching).

Preliminary analysis showed a significant association between prey size and male status which differed for different phases of the breeding season (analysis of 3-way contingency table: $G = 7.81$, df 3, $P < 0.05$). Overall, there was a significantly greater proportion of large prey delivered by polygynous males than by monogynous males after hatching ($G = 8.0$, df 1, $P = 0.005$), but not during courtship and incubation. The proportion of large prey delivered by females was lower than by polygynous males and higher than by monogynous males, but did not differ significantly from either.

There were some slight differences between individuals in the size of prey taken. For further analysis, I combined courtship with incubation and nestling with post-fledging phases, but even so some samples were small: I therefore omitted birds for which the sample was less than 20 for either of the combined phases.

There was no significant difference between two polygynous males and one monogynous male before hatching, or for the latter before and after hatching. The two polygynous males, however, took a significantly greater proportion of large prey after hatching than before ($G = 9.98$, df 3, $P < 0.025$). After hatching, two monogynous males differed significantly in the proportion of large prey taken ($G = 10.87$; df 1, $P < 0.005$), as did five females ($G = 12.65$, df 4, $P < 0.025$), but three polygynous males did not.

Seasonal changes in prey size could also be examined using the estimated weights of identified prey, but since young birds, which formed a substantial proportion of the diet, could not be categorized as pulli or juveniles in pellets or remains, the analysis was confined to sightings.

The females provided too few data for meaningful analysis and differences between individual males were too slight to be clear from the data available. However, the mean weight of prey delivered by polygynous males

Table 4. Seasonal variation in mean prey weights (g) from sightings of male and female Marsh Harrier prey with standard errors and sample sizes

	April	*May*	*June*	*July/August*
Polygynous males	70.2	107.6	143.1	183.7
	(21.0, 20)	(12.4, 63)	(12.1, 64)	(14.9, 76)
Monogynous males	20.0	116.3	135.9	124.6
	(0, 1)	(26.4, 13)	(18.6, 30)	(19.3, 33)
All males	67.8	109.1	138.4	161.6
	(20.1, 22)	(11.2, 76)	(10.1, 94)	(12.1, 109)
Females	299.0	—	149.5	198.0
	(0, 1)	(—, 0)	(39.2, 12)	(37.0, 7)

Notes: Correlations of weight with date: polygynous males, $r = 0.34$, df 222, $P < 0.001$; monogynous males, $r = 0.03$, df = 76, NS; difference between correlations, $t_s = 2.3$, $P < 0.05$. Difference between polygynous and monogynous males in July and August, $t_s = 2.3$, df 107, $P < 0.05$.

increased significantly during the summer, while that delivered by monogynous males did not, so that there was a clear difference apparent in July and August (Table 4).

The conclusion is that, while there were no differences between males of different status in the size of prey taken before hatching, polygynous males took larger prey items than monogynous males after hatching.

It seemed possible that the proportion of large prey items delivered by males could have been linked to brood size, which varied between 1 and 5 chicks. This was tested for the size class data for twelve individual nests but no significant correlation emerged ($r = -0.18$, df 10, NS). Mean prey weights from sightings were also tested against brood size for 7 males, but there was no significant correlation ($r = -0.02$, df 5, NS).

Prey categories

Table 5 shows the percentage of each major prey category from all birds in pellets, remains and sightings for different months. Those items in sightings which were classed by size but not identified have been included. The greatest contribution to χ^2 in all three samples came from a higher than expected proportion of mammals in June (April/May in sightings). The lower proportion of mammals later in the season in remains was largely due to a reduction in the contribution of Rabbits and in sightings and pellets a reduction in small mammals.

Since prey sightings could be attributed to particular birds, these data could be further tested for differences between polygynous and monogynous males, since preliminary analysis suggested a significant association between male status and categories of prey taken which differed seasonally (analysis of 3-way contingency table: $G = 33.98$, df 6, $P = 0.005$).

There were no significant overall differences between monogynous and polygynous males, nor between individual males within each category. (This was true whether one considered all males or, to reduce the variability caused by small samples, just the two in each category that provided large samples.) There was no significant variation between months in the proportion of the various prey categories taken by polygynous males (either for all of them or for the two large-sample individuals) but there was such variation for monogynous males (all males, $F = 13.0$, df 3,6, $P < 0.01$; large-sample males, $F = 15.3$, df 3,6, $P < 0.01$): these took fewer mammals and more young gamebirds—waterbirds later in the season.

These results suggest that the significantly greater proportion of larger prey taken by polygynous males after hatching is not due to a switch to different prey species but to taking larger individuals of the same species.

The only prey category which observers were able to categorize by size were young gamechicks and waterbirds, which were recorded as pulli or juveniles depending on whether they were less or more than one-

Table 5. Percentage of different prey types in the diet of Marsh Harriers in different months

	June	*July*	*August*
Pellets			
Small birds	27.5	44.9	31.8
Young gamebirds-waterbirds	30.3	31.4	31.8
Mammals	35.8	19.6	29.6
Other	6.4	4.1	6.8
Total prey items	109	194	88
	Mcy/June	*July*	*August*
Prey remains			
Small birds	22.3	32.9	26.4
Young gamebirds-waterbirds	20.2	40.9	43.4
Mammals	42.6	18.8	17.0
Other	14.9	7.4	13.2
Total prey items	94	149	53
	April/ May	*June*	*July/ August*
Sightings			
Small birds	11.7	2.8	6.7
Young gamebirds-waterbirds	9.2	12.9	16.2
Mammals	26.5	6.6	3.9
Other and unidentified	52.6	77.7	73.2
Total prey items	196	457	414

Differences between months: pellets $\chi^2 = 15.01$, df 6, $P < 0.025$; prey remains $\chi^2 = 28.87$, df 6, $P < 0.005$; sightings $\chi^2 = 116.54$, df 6, $P < 0.005$.

quarter grown. Table 6 shows the numbers in each category delivered by monogynous males, polygynous males, and females during May–August.

Table 6. Sightings of partly grown young gamebirds and waterbirds carried by Marsh Harriers

	May	*June*	*July*	*August*
Monogynous males				
pullus	2	10	3	0
juvenile	0	5	9	2
Polygynous males				
pullus	15	28	20	0
juvenile	1	10	26	0
Females				
pullus	0	6	2	1
juvenile	0	0	1	3

Note: Birds were classed as pullus or juvenile according to whether they were less or more than one-quarter grown.

The trend towards taking more juveniles as the season progresses (which would be expected to occur anyway, since there are more juvenile birds and fewer pulli available as the season progresses) is far stronger for polygynous males ($Z = 3.95$, $P < 0.001$) than for monogynous males ($Z = 2.13$, $P = 0.03$; Z test for trend in proportions), although overall the monogynous males took a significantly higher proportion of juveniles (51%) than polygynous males (37%) (normal approximation to binomial: $Z = 2.21$, $P = 0.03$).

DISCUSSION

Several types of bias could have affected the results. A bias in sightings towards larger items and some mammals was probable, as these were more easily identified (the tails of rodents and mustelids often hung down), whereas small items were normally completely hidden by the feet. This bias was reduced by considering items seen to be killed which were later identified, and by including items seen and classed but not identified, in all calculations involving sightings, but is nevertheless probably the reason for the significantly greater proportion of large prey among sighted items (those that could be identified) than among items that could only be placed in size classes. This does not seem to have affected the seasonal trends detected, since these were similar in both samples.

Rabbits and Pheasant poults were probably over-estimated as a proportion of the diet in pellets, since these were made up largely of fur and small feathers, with those of young Pheasants being particularly distinctive. Harrier digestion seems to be very thorough, since bones of prey species were largely digested if swallowed, as few appeared in pellets. The data from pellets are thus probably the least reliable, due to differential survival rates of identifiable remains (Yalden & Yalden 1985). All prey remains found were removed, and regular searches were undertaken of the same places, so this sample is likely to be least biassed towards larger items. It is unlikely that prey remains were from other predators, since these were all gathered from areas which were close to the nest sites, where the Harriers were seen to take prey for plucking or eating.

Small mammals could have been under-represented in remains since these were probably swallowed whole. Eggs, which occurred in pellets as pieces of shell were not seen being eaten or carried, and were found only once in remains.

Despite these difficulties, there was remarkably good agreement between the three sets of data for the main prey categories, with less than a 10% difference between all three for gamebirds, waterbirds and mammals, and 16% (between pellets and sightings) for small birds.

Practically all species of bird and mammal to be found within the Harrier's hunting ranges, and of an appropriate size, were represented in the diet. The only exceptions were species found only in woodland where Harriers did not hunt.

Seasonal changes in the composition of the diet were partly related to the availability of young gamebirds and waterbirds. Small adult birds and mammals formed the bulk of the food early in the season but as young birds appeared, relatively fewer mammals were taken. The differences between months for small birds in pellets and prey remains was largely due to an influx of young Starlings in July. Observations of hunting birds suggested that the Harriers changed their foraging behaviour by concentrating on this new food source, particularly in 1983 when the Starlings were present in large numbers. House Sparrows were also more numerous after July, when post-breeding flocks appeared in the fields.

The suggestion that the decline in the British Marsh Harrier populations during th t late 1950s and early 1960s was probably attributable to organo-chlorines (Underhill-Day 1984) is consistent with the finding that small birds figure prominently in the diet in early spring, since it was at this time of year that they were most vulnerable from dieldrin used as a seed dressing on spring planted grain (Cooke, Bell & Haas 1982).

Young gamebirds and waterbirds appeared in the diet from May onwards and continued to form an important constituent until August, with Harriers taking the young birds until the latter were about half-grown. Few large adult birds were recorded as prey and although Harriers were quite often observed stooping at adult Pheasants and Wood Pigeons, very few were attacked; the only Wood Pigeon recorded as prey had been pricked by gunshot earlier in the day. Female Pheasants and Partridges defended their broods, and on numerous occasions drove off a hovering Harrier by jumping up at it. Of 29 occasions when Harriers were seen to attack gamebird broods, they were successful on 14, usually when the brood scattered.

Because courtship feeding by the males before laying took place over a wide area and was difficult to observe, it was not possible to examine whether polygynous males provided larger prey to their females than monogynous males at this time, but they did not supply larger prey during the incubation period. All males took larger prey when feeding young than before hatching but this trend was significant only in polygynous males. Prey taken by polygynous and monogynous males were unlikely to reflect differences in spectrum or density of prey species within individual hunting ranges since these overlapped and covered an area of habitat with a high degree of homogeneity. Seasonal changes in the categories of prey taken by monogynous males together with the variation in proportion of large prey between individual birds may have been due to greater opportunism or inexperience. It was possible that older, more experienced males were more likely to be polygamous.

Mean brood size for five monogynous males was 3.6 and for three polygynous males was 5; the latter also had 2.2 and 3 females. Although there were no correlations between the size of prey delivered and brood size, these calculations took no account of the extent to which each female was provisioned by the male, which varied considerably between females. The finding that the polygynous males brought a higher proportion of larger prey than the monogynous males accords with a similar finding in the Netherlands (Altenburg *et al.* 1982), whilst the finding that the prey taken by females was larger than that taken by monogynous males is consistent with studies of other sexually dimorphic birds of prey (Newton & Marquiss 1982; Opdam 1975; Schipper 1973). Polygynous males brought at least the same proportion of large prey as the females, which is surprising in view of the much larger size of the latter.

In the study area captive-reared Pheasants were not released until late in the season and no Partridges were released. Most of the young gamebirds were taken during June and July from the substantial wild populations. Where Harriers are breeding, gamebirds that are reared for release can be largely protected by delaying their release until early August where this should be done in areas where there is plenty of tree cover. In this way few young birds should be taken.

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Impact of an exceptional winter flood on the population dynamics of bearded tits (*Panurus biarmicus*)

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Abstract

Mark–recapture data are used to investigate the impact of an exceptional winter flood on the population dynamics of an isolated bearded tit *Panurus biarmicus* population in north-west England. Adult numbers increased threefold (from 60 to 180 individuals) between 1992 and 2000, but declined by 94% during 2000/2001. This large reduction in numbers was caused by severe over-winter mortality associated with a prolonged flood of the *Phragmites* litter layer, the main winter foraging habitat of bearded tits, followed immediately by cold weather. At the end of the flood, bearded tits were 20% lighter than during previous winters. Population changes in other years were accurately predicted by annual variation in recruitment, and there was evidence that recruitment was higher following the introduction of artificial nest boxes and was density dependent. This study highlights the potential threat to bearded tits in Europe of predicted increases in the frequency and extent of winter flooding, and the potential utility of measures like artificial nest boxes that aim to promote recruitment. Recent extensions of methodology now permit a detailed exploration of animal population dynamics from mark–recapture data alone.

Introduction

Increases in the observed and predicted frequency and severity of extreme weather events is raising concern about the impacts of such events on biodiversity and ecosystem functioning (e.g. Easterling *et al*., 2000; Parmesan, Root & Willig, 2000; Diaz-Delgado *et al*., 2003; Ciais *et al*., 2005). In north-west Europe, long-term weather records highlight a trend for drier, warmer summers and substantially wetter autumns and winters with increased frequency of heavy precipitation events and flooding (Osborn & Hulme, 2002; Fowler & Kilsby, 2003), and many predictive scenarios forecast these trends to continue (Hulme, Turnpenny & Jenkins, 2002; Ekstrom *et al*., 2005).

Current knowledge of the impacts of extreme weather events on animal populations is sparse, partly because extreme events are by definition rare. Impacts of summer drought and winter freezing of pastures have been demonstrated for mammals (Garel *et al*., 2004; Chan *et al*., 2005), and a severe hurricane had no impact on deer in France (Gaillard *et al*., 2003). Documented impacts of extreme events on birds include severe winter weather (North, 1979; Marchant *et al*., 1990), summer rainstorms (Smith & Webster, 1955), habitat disturbance caused by a severe ice storm (Jones *et al*., 2001) and droughts in arid regions (Peach, Baillie & Underhill, 1991). Improving our understanding of the impacts of extreme weather events on wildlife is vital if potential mitigation measures are to be developed and implemented.

In this study, we use modern capture–mark–recapture (CMR) methods to diagnose the demographic and environmental causes of a near-extinction event affecting an isolated population of bearded tits *Panurus biarmicus* in north-west England. We also identify the main demographic driver of population changes in years before and after the near-extinction event. Bearded tits are small resident passerines whose distribution is strongly associated with *Phragmites* reed beds. They are widely distributed across northern and western Europe, but are restricted in Britain to larger reed beds mainly in southern and eastern England (Campbell, Cayford & Pearson, 1996). Bearded tits were scarce in Britain before a marked population expansion in the mid-1960s, but numbers had increased to 339–408 pairs by 1992 (Campbell *et al*., 1996). Bearded tits are included on the UK's amber list of Birds of Conservation Concern as a consequence of their population being restricted to a small number of breeding sites (Gregory *et al*., 2002).

Methods

Study area and field techniques

Since 1992, bearded tits have been the subject of a standardized CMR study at Leighton Moss Nature Reserve, in

Lancashire, north-west England (grid reference SD482750). The study area comprised 79 ha of *Phragmites* reed bed dissected by a network of dykes. Most of the reed bed stands in 10–20 cm of water, with limited drier areas along the margins. Leighton Moss was first colonized by bearded tits in 1973, numbers rising to about 40 pairs by 1980 and declining to about 12 pairs in 1987 (Wilson, 1993).

Bearded tits are difficult to census because they nest in loose colonies and are not overtly territorial, and because wet reed beds are often inaccessible. An annual census of first nesting attempts (late March to late April) based on intensive observations of breeding adults was undertaken at Leighton Moss between 1992 and 2004 (Wilson, 1993). Most of this census work was undertaken from access paths or using boats. From 1997 to 2004, artificial reed 'nest boxes' were provided and used by up to 60% of the population for first nesting attempts of the year. As vegetation cover increases during the spring, there is a partial shift away from nesting in artificial boxes to natural sites for later broods.

Bearded tits were captured in six 120 m mist-net rides distributed throughout the reed bed. Mist nets were erected in each of these rides for 4–6 h on two to three mornings each month between June and November 1992–2004. Each bird was fitted with a uniquely numbered aluminium ring and assigned to age (adult or first year) and sex classes following Svensson (1992). Of 1541 individual bearded tits captured during 1992–2003, 442 (29%) were recaptured during subsequent years. Most were first captured as juveniles (790 males and 688 females), with few first being captured as adults (39 males and 24 females) especially after 1992 (15 and six, respectively).

Estimating survival and recruitment rates

We estimated annual survival rates from between-year recaptures using standard Cormack–Jolly–Seber (CJS) models (Lebreton *et al*., 1992). Our analysis recognized four groups of birds (adults and first-years, males and females), and our starting model was one in which both survival and recapture rates differed between years and groups. Goodness-of-fit tests from program RELEASE (Lebreton *et al*., 1992) indicated that this model provided an adequate description of the data (all groups TEST 3: $\chi^2 = 42.3$, d.f. = 46, $P = 0.63$; all groups TEST 2: $\chi^2 = 22.1$, d.f. = 18, $P = 0.22$), and the data were not overdispersed ($\sum \chi^2/\text{d.f.} = \hat{c} = 1.01$). Thus, there was no evidence that the assumptions of the CJS were violated. We fitted a variety of simplified models (dropping age, sex or year dependence in survival or recapture parameters), and we used the Akaike's information criterion adjusted for small sample size (AIC_c) to identify the most parsimonious descriptions of the data and likelihood ratio tests (LRTs) to assess specific hypotheses (Lebreton *et al*., 1992; White & Burnham, 1999).

We tested whether interannual variation in survival was related to the severity of winter (November–February) weather conditions. We tested two measures of winter temperature (the mean daily temperature of the coldest month and the number of continuous days when minimum grass temperatures fell below −5 °C) as well as the longest period (in days) of continuous snow cover (i.e. snow ground cover at 09:00 GMT). Daily temperature records were taken from a weather station (Leven's Hall) 9 km to the north of the study area. We also tested for a possible effect of winter flooding of reed litter on survival. During November–January the diet of bearded tits is dominated by reed seeds (>70% of items in faecal remains), with invertebrates predominant during all other months (Bibby, 1981). Observations at Leighton Moss since 1973 indicate that reed litter is the preferred foraging habitat of bearded tits when gathering reed seed, with only occasional foraging on standing reed heads during periods of snow cover or heavy frost (see Discussion). Thus, flooding of reed litter during winter probably renders the majority of the preferred foraging habitat unavailable. We tested whether interannual variation in survival was related to the number, and longest duration, of days each winter when water levels at Leighton Moss exceeded 85 cm, which is the level at which most of the reed litter floods. We also tested whether survival was density dependent. We estimated the strength of covariate relationships by calculating the proportion of temporal (process) variation accounted for by covariates (Loison *et al*., 2002). This assumes temporal variation in demographic rates to be a random process and involves decomposing total variance into sampling and process (or 'true') variance (Burnham & White, 2002).

We used 1345 body mass measurements collected during October–January during six winters (1995/1996 to 2000/2001) to investigate the timing and possible causes of the severe mortality that occurred during the 2000/2001 winter. A generalized linear model (GLM) was used to assess the influence of age–sex status (four-level factor), winter (six-level factor), date and date^2 (days since 1 October; both continuous covariates) and associated interactions on winter body mass. Because there was no significant relationship between body mass and wing length ($P = 0.34$) after significant differences in body mass between age–sex classes had been accounted for, wing length was not included as a covariate.

We used reverse-time CMR methods to estimate annual recruitment to the population (Pradel, 1996). Specifically, we estimate fecundity rates, which are the number of recruits to the population per animal in the previous year (Williams, Nichols & Conroy, 2001). This analysis was restricted to adult data (birds initially captured as first-years only entered the analysis if recaptured as adults), and involved testing for effects of year and sex on survival, recapture and fecundity rates. We test whether fecundity was density dependent or related to the presence/absence of artificial nest boxes (introduced in 1997 and present every year thereafter).

Estimating adult abundance

We used Pollock's robust design (RD) to estimate adult population size using same-year recaptures (Pollock, 1982; Pollock *et al*., 1990; Kendall, Nichols & Hines, 1997). The

RD involves combining open and closed CMR models within a single likelihood function, to estimate survival rates between successive 'primary' sampling periods (year, in this case) and population size within primary periods using recaptures between 'secondary' sampling periods within each primary period (Pollock, 1982). Our analysis is based on three secondary periods: May–June, July–August and September–October. In addition to survival and population size estimates, Kendall *et al.*'s (1997) RD implementation incorporates capture and recapture rates and temporary emigration parameters. Leighton Moss is an isolated reed bed with only one significant patch of reed bed within 25 km. This 10 ha patch is 1 km from Leighton Moss, but contains extensive willow scrub (which is avoided by nesting bearded tits) and frequent annual checks have produced no summer observations of bearded tits at this site (R. P. Jones, pers. comm.). We therefore assumed no temporary emigration of bearded tits from our study population and fixed the temporary emigration parameter to zero.

Because the CJS analysis indicated significant interannual variation in adult survival (see Results), all our RD models retained annual variation in survival, plus year- and sex-specific population size parameters. Since CJS recapture rates did not differ significantly between years (see Results) and capture effort was standardized across years, we also constrained RD capture and recapture rates to be equal across years. LRTs indicated that it was necessary to allow capture and recapture rates to differ between the sexes and between secondary sampling periods (i.e. three capture and two recapture parameters for each sex). However, several of the population size estimates were not estimable in models in which all of these capture rates were estimated separately. Capture and recapture estimates increased across the three secondary periods, and by constraining each to change linearly and independently across secondary periods we were able to estimate population sizes for both sexes in all years.

By allowing capture rates to be equal across primary periods (years), the RD offers large gains in precision compared with year-specific closed population models of the sort described by Otis *et al.* (1978). We checked that this gain in precision did not introduce bias into population estimates by comparing RD estimates with those from closed population models fitted separately to each year's data. For the latter we used model 'Mt' of Otis *et al.* (1978), which allows for temporal variation in capture probabilities across secondary sampling periods (as was needed with our data). Adult population estimates were on average 8% higher under the RD; however, the two sets of estimates were strongly correlated ($R^2 = 0.97$), suggesting that the additional assumptions of the RD did not seriously bias population estimates.

All CJS, recruitment and RD models were fitted using software MARK 4.1 (White & Burnham, 1999).

Productivity

We used the number of young birds in the annual catch (June–November) divided by the number of occupied nests (from the annual survey) as an index of annual productivity that integrates average nesting success, the average number of nesting attempts and post-fledging mortality. A positive relationship between this index and the mean number of young fledged per nesting attempt during 6 years when nest monitoring was conducted (1998–2001, 2003 and 2004) provides some independent validation of the index (Fig. 1). Although the relationship is not formally significant (Pearson's correlation weighted by the reciprocal of the standard error of the number of fledglings per nest: $r = 0.62$, $P = 0.19$), it becomes significant if the 2001 datum (highest fledglings per nest estimate) is excluded ($r = 0.93$, $P < 0.03$).

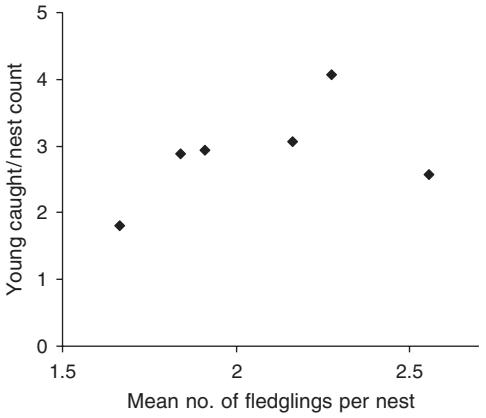


Figure 1 Relationship between the numbers of young bearded tits *Panurus biarmicus* caught in mist nets each year and the mean number of young fledging from nests monitored during 6 years.

Demographic drivers of population change

We assessed the sensitivity of the population multiplication rate (PMR = N_{i+1}/N_i, where N_i is the population size in year i, based on RD population estimates) to temporal variation in the various demographic components by testing for correlations between the observed PMR and that predicted by a Leslie matrix model (Caswell, 2001) in which only one demographic rate was allowed to vary annually, and all birds were assumed to breed when 1 year old. Because the population decline of 2001 was clearly driven by exceptionally low winter survival during the preceding winter, we included year-specific survival estimates for 2000/2001 and average rates for all the other years. Thus, the influence of temporal variation in first-year survival on PMR was assessed by holding productivity constant and adult survival constant for all years except 2000/2001, and allowing first-year survival to vary annually. Productivity was estimated iteratively as the constant that minimized squared deviations between observed and predicted population sizes. When assessing the influence of annual variation in productivity on PMR, the average value of first-year survival was estimated iteratively in the same way. The influence of temporal variation in fecundity on PMR was estimated by summing the annual fecundity estimates with

the average adult survival estimate for all years except 2000–2001 when the year-specific value was used (Williams *et al.*, 2001).

Results

Population size

Nest counts and RD estimates indicate a gradual increase in the number of adult bearded tits until 2000, followed by a sudden and large decline in 2001 (89% of nest counts, 94% of RD estimates) and a subsequent partial recovery (Fig. 2). Only seven pairs were located in 2001 and the CMR data suggested between nine and 27 adults (95% confidence limits).

During most years of the study, RD estimates of adult abundance were similar to nest counts with a slight excess of males. During 1998–2000, estimates of male abundance considerably exceeded those of females (males accounting for 67, 70 and 60% of adults, respectively) and nest counts (Fig. 2). RD estimates of female numbers were similar to nest counts in most years (Fig. 2) and the two were strongly correlated across years ($r = 0.94$, $n = 13$, $P < 0.0001$). RD estimates of abundance were insensitive to the inclusion of year-specific capture and recapture rates for the years 1998–2000.

Survival

Recapture rates differed between males and females and between adults and first-years, but not between years (Table 1). There was strong evidence of interannual variation in apparent survival and of higher average survival among adult bearded tits. The pattern of interannual variation in survival was similar for adults and first-years, and a model in which annual variation in survival was additive (on a logit scale) across age classes was preferred to one in which the temporal pattern of variation differed between age classes (LRT between models 2 and 5 in Table 1: $\chi^2 = 18.6$, d.f. = 11, $P < 0.07$). Apparent survival was exceptionally low among adult and first-year bearded tits during 2000–2001 (Fig. 3), and a relatively parsimonious description of the data was provided by a model in which first-year survival differed between years while adult survival differed only between the exceptional mortality year (2000–2001) and all other years (0.036 vs. 0.558; model 1 in Table 1). The high mortality during 2000–2001 coincided with the large decline in adult abundance during 2001 (Fig. 3). The demographic basis of the temporal changes in adult sex ratio is considered in Supplementary Material Appendix S1.

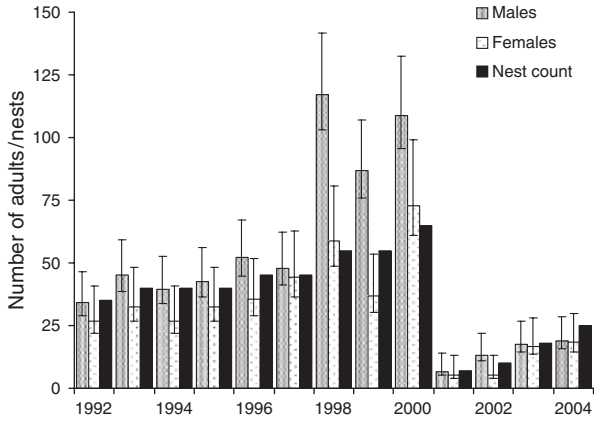


Figure 2 Temporal changes in the number of active nests and in the number of adult bearded tits *Panurus biarmicus* (with 95% confidence intervals) estimated from robust design capture–mark–recapture models.

While all four winter weather and flooding variables accounted for significant amounts of interannual variation in survival, the number of days the reed litter was flooded each winter was the strongest predictor (Table 1; LRT between models 13 and 16: $\chi^2 = 290.8$, d.f. = 2, $P < 0.0001$). The relationship between survival and winter flood days suggests a non-linear effect whereby survival increases up to some threshold amount of winter flooding, beyond which survival falls dramatically (Fig. 4a). After allowing for the non-linear relationship between survival and flood days, there was a weak negative influence of the duration of cold winter temperatures (LRT between models 14 and 16: $\chi^2 = 3.7$, d.f. = 1, $P < 0.06$) but no evidence that survival was density dependent (LRT between models 16 and 18: $\chi^2 = 1.0$, d.f. = 1, $P > 0.3$).

Winter body mass

Mean body mass differed significantly between age–sex classes and winters, and the pattern of temporal variation differed significantly between winters (Table 2, Fig. 5a). During all winters except 2000–2001, body mass increased during November and December, with most birds weighing 15–20 g during December and January (Fig. 5a). In 2000–2001, all 12 birds captured during the period 26 December (day 87) to 9 January (day 101) weighed less than 15 g (Fig. 5a), and 11 of these were never captured again.

Average bearded tit weights were less than 1 g lighter during October 2000 compared with the previous five Octobers, but were 3–4 g (*c.* 20%) lighter by late December 2000 than during previous winters (Fig. 5a). The flooding of the reed bed had receded by 23 December 2000 (day 84, Fig. 5b), but was immediately followed by nine days of cold conditions (days 84–92) including 4 days of continuous snow cover (days 89–92), with 14 further days of cold weather soon after (days 100–114). The exceptionally low weights recorded on 26 December 2000 (day 87) occurred after only 3 days of cold weather and shortly after more than 50 days of almost continuous flooding of the reed litter (Fig. 5b).

There was no evidence that mean wing length or body mass (measured during October or November 2000) differed between the few birds that survived the 2000–2001 winter and those that did not (ANOVA after allowing for

Table 1 Modelling apparent survival (Φ) and recapture rates (P) of bearded tits *Panurus biarmicus*

	AIC_c	ΔAIC_c	AIC_c weights	No. of parameters	LRTs (covariate and significance)	% of process variance in $\Phi(a \times t)$	
						Adult	First-year
Model[a]							
1. Φ(ADfl, FYt) $P(a \times s)$	2852.2	0	0.987	18			
2. $\Phi(a + t)$, $P(a \times s)$	2861.1	8.9	0.012	17			
3. $\Phi(a \times t)$, $P(a \times s)$	2865.0	12.8	0.002	28			
4. $\Phi(t)$, $P(a \times s)$	2882.3	30.2	0	16			
5. $\Phi(a \times t \times s)$, $P(a \times s)$	2894.2	42.0	0	52			
6. $\Phi(a \times t \times s)$, $P(a)$	2897.3	45.1	0	50			
7. $\Phi(a \times t \times s)$, $P(s)$	2903.7	51.5	0	50			
8. Φ(ADfl, FYfl) $P(a \times s)$	2921.2	69.0	0	8			
9. Φ(ADt, FYfl) $P(a \times s)$	2931.6	79.4	0	18			
10. $\Phi(a \times t \times s)$, $P(a \times t \times s)$	2936.2	84.0	0	92			
11. Φ(ADc, FYt), $P(a \times s)$	2971.6	119.4	0	17			
12. Φ(ADt, FYc), $P(a \times s)$	3096.9	244.7	0	17			
13. $\Phi(a)$, $P(a \times s)$	3198.9	346.7	0	6			
Covariate models[b]							
14. $\Phi t = a + WD + WD^2 + T5$	2910.4	58.2	0	9	$T5^{*}$	100	36
15. $\Phi t = a + WD + WD^2 + T5 + ABU$	2911.8	59.6	0	10	ABU^{NS}	100	30
16. $\Phi t = a + WD + WD^2$	2912.1	59.9	0	8	$WD^{2,***}$	100	36
17. $\Phi t = a + WD + WD^2 + T_{min}$	2913.6	61.4	0	9	$T_{min}{}^{NS}$	100	27
18. $\Phi t = a + WD + WD^2 + ABU$	2913.7	61.5	0	9	ABU^{NS}	100	27
19. $\Phi t = a + WD$	2991.4	139.2	0	7	WD^{***}	58	19
20. $\Phi t = a + WC + WC^2$	3014.6	162.4	0	8	$WC^{2,***}$	57	19
21. $\Phi t = a + WC$	3104.7	252.5	0	7	WC^{***}	6	0
22. $\Phi t = a + T5$	3142.0	289.8	0	7	$T5^{***}$	0	0
23. $\Phi t = a + SD$	3158.5	310.9	0	7	SD^{***}	0	0
24. $\Phi t = a + T_{min}$	3173.9	321.7	0	7	$T_{min}{}^{***}$	0	0

[a]Parameters can be age (a), sex (s), year (t) or flood year (fl) specific, and can differ between adult (AD) and first year (FY) age classes. Models are listed in descending order of parsimony as measured by AIC_c. Also listed are differences in AIC_c between each model and the best model (ΔAIC_c), the relative support that the data provide for each model (AIC_c weights) and the number of identifiable parameters. Likelihood ratio tests (LRTs) assess the significance of the last covariate included in the model, while model strength is assessed through the percentage of temporal process variance explained by all covariates in the model (see text).

[b]All covariate models include age-specific intercepts and a common slope across age classes. $T5$, longest run of days each winter when minimum temperatures remained below $-5\,^{\circ}C$; T_{min}, mean daily temperature of the coldest winter month; SD, longest continuous run of days with snow cover; WD, number of days each winter that reed-bed water depth exceeded 85 cm; WC, number of continuous days each winter that reed-bed water depth exceeded 85 cm; ABU, adult abundance (robust design estimates). Recapture rate (P) was always age- and sex-specific [i.e. $P(a \times s)$].

$^{*}P < 0.07$; $^{***}P < 0.001$; $^{NS}P > 0.10$.

AIC_c, Akaike's information criterion.

significant age and sex effects: $F_{1,80} = 2.0$, $P = 0.15$ for wing length, $F_{1,288} = 0.35$, $P = 0.55$ for mass).

Recruitment

As with the CJS analysis, adult recapture rates differed between the sexes, and adult survival was exceptionally low during the flood year but was otherwise relatively constant across years (Table 3). Fecundity (per capita recruitment rate) varied significantly between years (LRT between models 1 and 9 in Table 3: $\chi^2 = 127.5$, d.f. = 10, $P < 0.0001$). Fecundity was low following the flood of 2000 (Fig. 6), was higher following the introduction of artificial nest boxes in 1997 (LRT between models 14 and 15 in Table 3: $\chi^2 = 11.9$, d.f. = 1, $P < 0.001$) and, having allowed for this effect, was negatively density dependent (LRT comparing models 13 and 14: $\chi^2 = 10.9$, d.f. = 1, $P < 0.001$; Fig. 7).

Productivity index

The number of young birds caught per active nest located varied between 1.8 (in 1998) and 4.1 (in 2000), showed no evidence of being density dependent (correlation with RD adult abundance estimates for the same year: $r < 0.01$, d.f. = 11, $P > 0.9$), and showed no obvious association with adult population size in the following year (Fig. 6b). The index was uncorrelated with the proportion of all nests in nest boxes ($r_{11} = 0.13$, $P = 0.65$) and did not differ between years when nest boxes were present and absent (two-sample median test: $Z = -0.34$, $P = 0.73$).

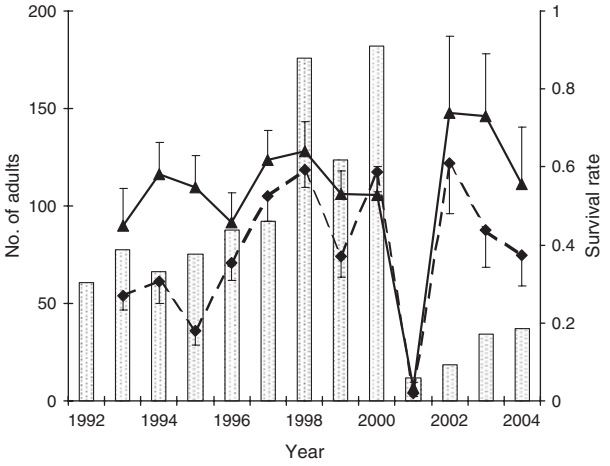


Figure 3 Temporal changes in adult (triangles) and first-year (diamonds) survival rates (± SE) in relation to changes in the abundance of adult bearded tits *Panurus biarmicus* (estimated from robust design models).

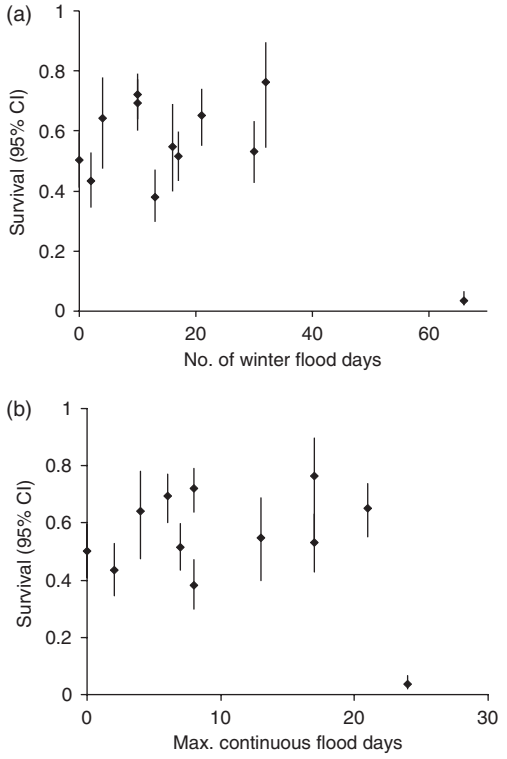


Figure 4 Relationship between bearded tit *Panurus biarmicus* survival (for both adults and first years, taken from model 2, Table 1) and (a) the number of days each winter (November–February) when water levels were high enough to flood the reed litter and (b) the maximum duration of flooding each winter. CI, confidence interval.

Table 2 Factors affecting the body mass of bearded tits *Panurus biarmicus* captured between October–January 1995–1996 and 2000–2001 at Leighton Moss

Factor or covariate and significance	Direction of effects	Partial R^2 (%)
Age–sex[a,***]	AdM > FyM > AdF > FyF	13.3
Winter[b,***]	4 > 2 > 3 > 1 > 5 > 6	21.4
Date	Positive	2.4
Date × age–sex***	AdM > AdF > FyF > FyM	0.9
Date × winter[b,***]	1 > 6 > 3 > 2 = 5 > 4	5.3
Date-squared***	Negative	0.6
Date-squared × winter[b,c,***]	4 > 5 = 2, 3 = 1 > 6	1.6

[a]Ad, adult; Fy, first-year; M, male; F, female.
[b]Indices 1–6 refer to winters 1995–1996 to 2000–2001, respectively.
[c]'Date-squared × winter' was no longer significant ($P = 0.069$) when winter 6 (2000–2001) was excluded.
***$P < 0.005$.

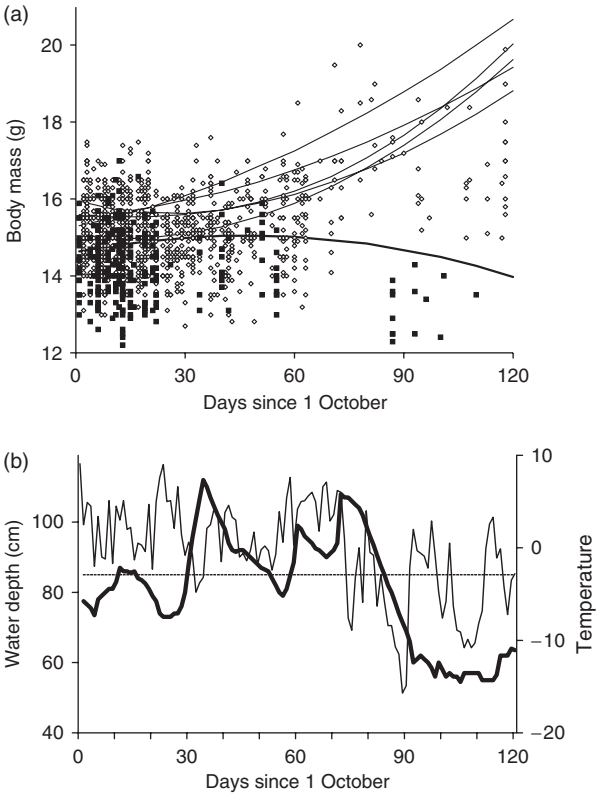


Figure 5 (a) Changes in bearded tit *Panurus biarmicus* body mass between October and January during the winters 1995/1996–1999/2000 (open diamonds) and 2000–2001 (filled squares). Trend lines show predicted changes in body mass of adult males for each winter (thin lines: 1995/1996–1999/2000; thick line: 2000–2001) as derived from a generalized linear model (see text). (b) Daily variation in water depth at Leighton Moss (thick line) and minimum grass temperature (thin line) during the 2000–2001 winter.

Demographic drivers of population change

Because exceptionally low over-winter survival was clearly the main demographic mechanism underlying the large population decline of 2001 (Figs 2 and 5), the key remaining question is which demographic parameter was the main driver of population change during other years. Annual

Table 3 Modelling survival (Φ), fecundity (*f*) and recapture (*P*) rates of adult bearded tits *Panurus biarmicus*

	AIC_c	ΔAIC_c	AIC_c weights	No. of parameters	LRTs (covariate, significance)	% of process variance in *f*(*t*)
Model[a]						
1. Φ(fl) *f*(*t*) *P*(*s*)	3989.0	0	0.960	15		
2. Φ(*t*) *f*(*t*) *P*(*s*)	3997.1	8.1	0.017	25		
3. Φ(*t*) *f*(*t*) *P*	3998.1	9.1	0.010	24		
4. Φ(*t*) *f*(*t*) *P*(*t*)	4005.0	16.0	0.000	35		
5. Φ(*t*) *f*(*t* × *s*) *P*(*s*)	4012.2	23.2	0.000	37		
6. Φ(fl) *f*(fl) *P*(*s*)	4097.8	108.8	0.000	6		
7. Φ *f*(*t*) *P*(*s*)	4115.6	126.6	0.000	13		
8. Φ(*s*) *f*(*t*) *P*(*s*)	4116.4	127.4	0.000	14		
9. Φ(*t*) *f* *P*(*s*)	4152.9	163.9	0.000	15		
10. Φ(*t*) *f*(*s*) *P*(*s*)	4154.6	165.6	0.000	16		
11. Φ(fl) *f* *P*(*s*)	4163.0	174.0	0.000	5		
12. Φ(fl) *f*(*s*) *P*(*s*)	4164.7	175.7	0.000	6		
Covariate models[b]						
13. *ft*=FL+BOX+ABU	3997.8	8.8	0.012	8	ABU***	64
14. *ft*=FL+BOX	4006.6	17.6	0.000	7	BOX***	45
15. *ft*=FL	4016.6	27.5	0.000	6	FL***	33
16. *ft*=FL+ABU	4017.4	28.4	0.000	7	ABU^{NS}	35

[a]Subscripts indicate year (*t*), sex (*s*) or flood year (fl, i.e. 2000–2001, cf. all other years) dependence of parameters. Column headings follow Table 1.

[b]Testing relationships between fecundity and the presence/absence of nest boxes (BOX) and the abundance of adult bearded tits (ABU) in the previous year. Because fecundity was exceptionally low following the flood of 2000, fecundity in that year was excluded from the above relationships through the inclusion of a flood (FL) covariate. In all covariate models, adult survival differed between the flood and all other years while recapture was sex-specific [(i.e. Φ(fl), *P*(*s*)].

***$P<0.001$.

AIC_c, Akaike's information criterion; NS, not significant.

variation in fecundity provides a remarkably accurate predictor of observed bearded tit population changes, but the same is not true of other demographic rates (Fig. 8). Annual variation in fecundity (or per capita recruitment) therefore appears to have been the main driver of bearded tit population changes in years other than the flood year of 2000–2001.

Discussion

Exceptional mortality of 2000–2001

Apparent mortality during the winter of 2000–2001 was exceptionally high among adult and first-year bearded tits (96% compared with an average of 44% in other years for adults, and 98% compared with 60% for first-years) and the population at Leighton Moss came close to extinction. Although bearded tits sometimes undertake irruptive movements during autumn and winter (Axell, 1966; O'Sullivan, 1976), none of the characteristic excited 'high flying' behaviour associated with such movements was noted during the 2000–2001 autumn/winter despite frequent visits to Leighton Moss. Nor were there any reports that winter of colour-ringed birds away from the site.

Between November and January, bearded tits become heavily reliant on reed seed as food (Bibby, 1981). Regular observations at Leighton Moss since 1973 indicate that the vast majority of reed seed is collected from fallen reed litter, with only occasional *in situ* feeding on flower heads mostly when snow or heavy frost renders the litter inaccessible (J. Wilson, pers. obs.). Bearded tits may be vulnerable to predation from sparrowhawks *Accipiter nisus* when feeding on reed flower heads; two were killed by hawks during a 2-h watch in December 2000 when snow covered the reed litter. Thus, the prolonged flooding of the reed bed during November and December 2000 removed most of the preferred foraging habitat.

Bearded tit weights had reached exceptionally low levels (83% of expected weights for this time of year) by 26 December 2000 (day 87 in Fig. 5a), which was just 3 days after the flood receded (Fig. 5). Although it is possible that cold weather during the 3 days before this sample contributed to these low weights, the absence of comparable weight loss during longer periods of colder weather in previous winters suggests that the flood of the reed bed was the main cause of the exceptionally low mid-winter weights.

The amount and quality of seed produced by reeds varies markedly between sites and years (McKee & Richards, 1996), but it seems unlikely that lack of seed production could have caused severe mortality of bearded tits during 2000–2001. Weather conditions at Leighton Moss during 2000 should have promoted pollination (August was not unusually wet) and seed production (September and October were wet and mild) (McKee & Richards, 1996). Thus,

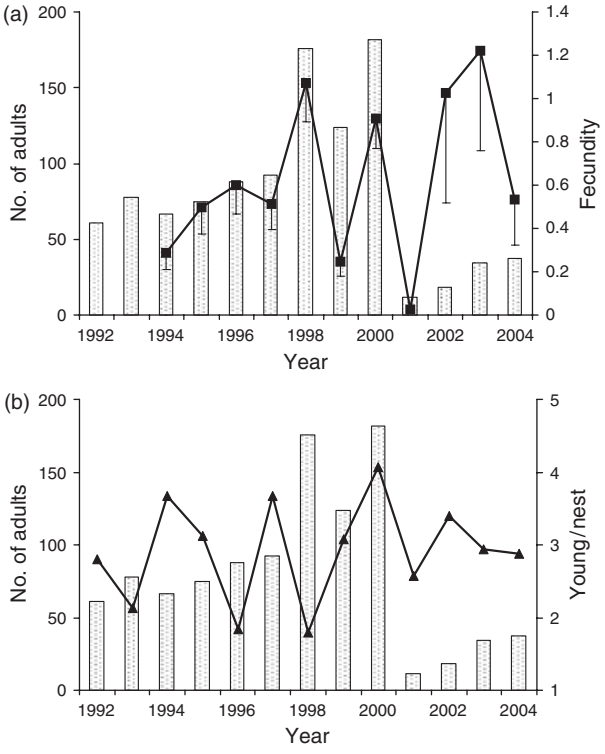


Figure 6 Temporal changes in (a) fecundity (± SE) and (b) the number of young birds captured per active nest in relation to changes in the abundance of adult bearded tits *Panurus biarmicus* (estimated from robust design models). Fecundity is an estimate of the number of new recruits to the population per adult present in the previous year.

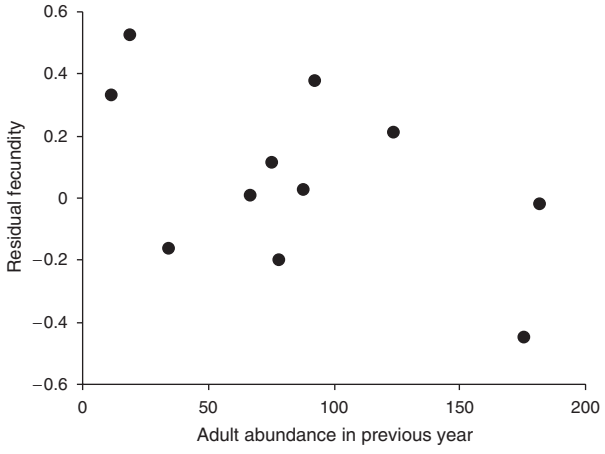


Figure 7 Relationship between annual fecundity (number of new recruits per adult in the population in the previous year) and the abundance of adult bearded tits *Panurus biarmicus* in the previous year. Residual fecundity is that after allowing for unusually low fecundity in 2001 (following the flood mortality) and higher fecundity after artificial next boxes were introduced.

the severe mortality of 2000–2001 appears to have been caused by a lack of available reed seed, which was primarily a consequence of the prolonged flooding of the reed litter and possibly exacerbated by the cold weather that immediately followed (Fig. 5b).

The flood of 2000 was an exceptional weather event. More rain fell during October–December 2000 in England and Wales than in any year since records began in 1766 (Environment Agency, 2001). More rain fell in our study area during September–December 2000 (787 mm) than during the same period in any of the previous 34 years for which we have data (range 222–675 mm, mean = 433 mm; rainfall exceeded 600 mm in only one other year, 1980). The weather that followed the flood was cold but not exceptional. A run of nine continuous days of minimum daily temperatures below −5 °C during January 2001 was shorter than equivalent cold runs during the 1993–1994 (14 days) and 2003–2004 (12 days) winters when bearded tit survival was normal (Fig. 3). The 4 days of continuous snow cover in late December 2000 were not exceptional and 6 continuous days of snow cover during December 1995 were not associated with unusual over-winter mortality (Fig. 3). Many global climate change scenarios predict increasingly wet autumns and winters in the UK and north-west Europe with extreme events becoming increasingly frequent (Hulme *et al*., 2002; Ekstrom *et al*., 2005), and the winter flooding of reed beds is likely to pose an increasing threat to bearded tit populations across Europe.

Population dynamics and bearded tit conservation

This study has shown that temporal variation in per capita recruitment has been the main demographic driver of bearded tit population changes (Fig. 8c) as well as a potentially regulating mechanism (Fig. 7). Until Pradel's (1996) recent extension of the CJS model, overall recruitment has rarely been measured for passerine bird populations because of methodological difficulties associated with measuring dispersal (Sutherland, Newton & Green, 2004). Most previous studies have focused on recruitment of locally bred animals only (Newton, 1998). Despite differences in definition, our data support the general conclusion of Saether & Bakke (2000) that avian population changes in relatively fecund, short-lived birds like bearded tits tend to be more sensitive to temporal variation in fecundity rather than adult survival.

As well as having a major impact on recruitment (via first-year mortality), the flood of 2000 also had a major negative impact on adult survival, a demographic parameter that was otherwise relatively invariable before and after the flood event. Moreover, the flood had a major influence on the 'true' temporal process variance associated with fecundity, first-year and adult survival (Supplementary Material Appendix S2). Such demographic sensitivity to an extreme event has important implications for techniques such as population viability analysis (PVA) that require realistic estimates of temporal demographic variance and catastrophic population declines of the sort described here (Coulson *et al*., 2001). A lack of knowledge of the catastrophic impact of flooding on bearded tits described here is

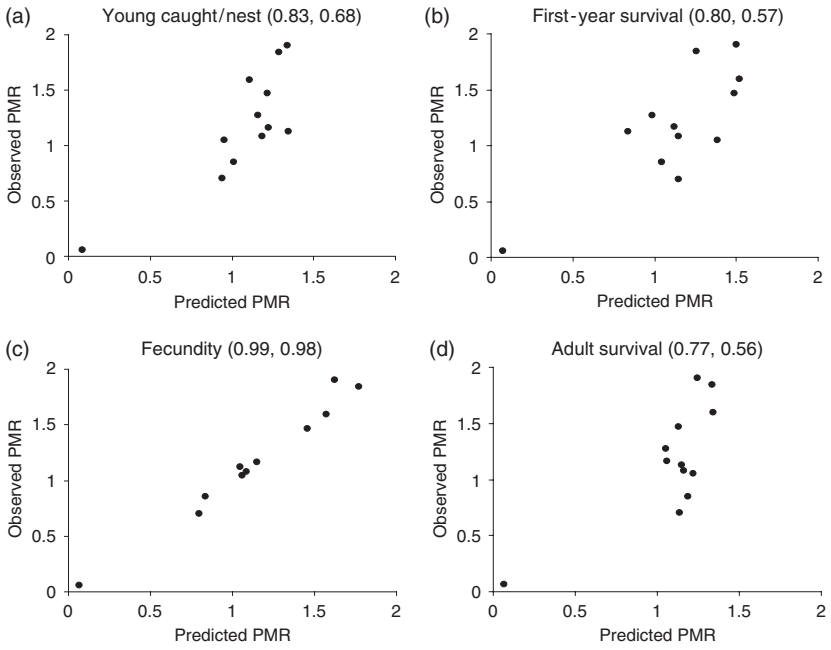


Figure 8 Relationship between observed annual population multiplication rate (PMR) and that predicted from a Leslie matrix model in which only one demographic rate was allowed to vary across years (see text). The time-variant rates are (a) the number of young tits caught each autumn divided by the number of active nests, (b) first-year survival, (c) fecundity and (d) adult survival. Observed PMR is derived from robust design abundance estimates, survival estimates are taken from model 3 of Table 1, and fecundity estimates are taken from model 1 of Table 3. Pearson's correlation coefficients are presented in parentheses for all years, and excluding the exceptionally low PMR of 2000/2001. Correlations between observed PMR (excluding 2000/2001) and the associated raw demographic parameters are: young/nest 0.68, $P<0.05$; first-year survival 0.57, NS; fecundity 0.98, $P<0.001$; adult survival 0.56, NS).

likely to result in serious underestimation of extinction probability based on PVA.

Our data suggest that annual recruitment increased following the introduction of artificial nest boxes in 1997. Bearded tits usually nest close to the ground in drier areas of reed beds among sedges or fallen reed stems, and fly to wetter areas to feed. The provision of artificial nest boxes at Leighton Moss has allowed bearded tits to nest in wetter areas of reed where they may be less vulnerable to predation or flooding and where they are closer to feeding areas (J. Wilson, pers. obs.). The possibility that nest boxes increase the survival of nests or fledglings warrants further study.

This study has clearly demonstrated the vulnerability of bearded tits to flooding of the reed litter during winter. At most reed beds, large areas of reed are cut during autumn or winter to prevent successional changes to the vegetation (Burgess *et al*., 1995). Much of this cutting takes place in the drier parts of reed beds where litter build-up is most advanced and where bearded tits prefer to forage especially when water levels are high. Other management techniques such as bed lowering and the raising of water levels have also reduced drier reed areas. Although such techniques promote the long-term health of reed beds and increase their attractiveness to bitterns *Botaurus stelliaris,* the importance of drier areas to bearded tits needs to be recognized in management plans. At sites where suitable dry reed areas are vulnerable to winter flooding, consideration should be given to leaving cut reed on ground that is unlikely to flood as a potential source of winter seed for bearded tits. Trials are needed to determine the best ways of storing and presenting the reed seed. Reed piles might also provide winter refuges for invertebrates and reptiles (Burgess *et al*., 1995). The provision of nest boxes looks promising as a technique to increase recruitment (and possibly breeding densities), but also requires further assessment.

Our study illustrates how modern analytical techniques allow the efficient estimation of population size, recruitment and survival rates from a single set of mark–recapture data. Standardization of capture effort across years ensured that recapture rates were constant across years and allowed the precision of key demographic parameters to be retained despite a small study population in some years. Pradel's (1996) recruitment rate combines productivity, survival from fledging to first breeding and immigration from outside the study area, all of which are difficult to measure using conventional approaches (Sutherland *et al*., 2004). These new methods have wide potential application in conservation biology and studies of population dynamics.

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Supplementary material

The following material is available for this article online:

Appendix S1 Demographic causes of changes in sex ratio.

Appendix S2 'Shrunken' estimates of survival and fecundity and associated temporal process variance derived from random effects models.

This material is available as part of the online article from http://www.blackwell-synergy.com/doi/abs/10.1111/j.1469-1795.2006.00063.x

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Józef WITKOWSKI

Breeding biology and ecology of the marsh harrier *Circus aeruginosus* in the Barycz valley, Poland

WITKOWSKI J. 1989. Breeding biology and ecology of the marsh harrier *Circus aeruginosus* in the Barycz valley, Poland. Acta orn. **25**: 223–320.

A population of marsh harriers was studied in 1972–75, when it was affected by DDT and shooting, and in 1982–84, seven years after the removal of both pressures. Since the end of the 19th century the population had two periods of growth (during World War II and after stopping the use of DDT) and a period of decline during most intense DDT application (1950–70). The breeding territory of a single pair ranged from 27 to 100 ha in both the study periods. The mean home range in the years of highest numbers was 160 ha. Only 4% of the females mated bigamously. They produced 40% less young than females mated monogamously. Bigynous males produced 30% more young than monogamous males. The average date of initiation of egg laying was 27 April. Females laid 2–8, on an average of 5.0 eggs. The mean clutch size declined from 5.6 eggs early in the season to 4.4 eggs late in the season. The first and the last eggs within a clutch were usually smaller than the eggs laid in between. No difference was recorded in the production of fledglings between clutches made up of larger eggs and those containing smaller eggs. The daily food intake by the young was 1/4 to 1/3 of their body weight. Males started to fly on days 37–39, whereas females on days 41–44 after hatching. Males, which are lighter than females, could catch little mobile prey when 54 days old, but they were still partly fed by parents up to 80 days of age. The sex ratio in the young was 54% in favour of males. Broods containing both sexes predominated but 26% of the broods contained only one sex. In 5-egg clutches, males hatched from relatively larger eggs, typically laid in the middle of the laying sequence, whereas females from smaller eggs laid at the beginning and at the end of the sequence.

In 1972–75, the highest losses were caused by egg breakage and low hatchability as a result of DDT contamination. The index of shell thickness in the Barycz valley was 23% lower than that of European marsh harriers prior to 1945. Losses from DDT were so high that they reversed the typical trend of lower losses at the beginning than at the end of the season. Prolonged rains additionally increased losses from egg breakage and nestling mortality. In 1982–84, the highest losses were due to predation and nest desertion. Nestling mortality accounted for 25%. Nestlings that hatched first survived until fledging in 94%. No one nestling hatched as the sixth survived. Aggression among siblings accounted for negligible losses. In 1972–75, 42% of the eggs laid produced young, whereas in 1982–84, 51%. The production of young in successful nests was the same in the two study periods (3.1 fledgings/pair). The marsh harrier population declined in the period of DDT application because of the elimination of some pairs from reproduction and as a result of the shortening of the life span of adults contaminated with DDT and its metabolites.

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The diet of marsh harriers in the breeding season (n = 1355 food items) consisted of birds – 60%, mammals, typically small rodents, – 28%, dead fish in 10%, and frogs – 2%. Males caught 58% of the prey items from land habitats and 24% from aquatic habitats, whereas females caught only 8% of the prey items from land and as many as 83% from aquatic habitats. A clear separation of food niches between males and females in the breeding season was evident within the population studied. On the average, the prey caught by the male was smaller by half than the prey caught by female.

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Биология и экология периода размножения болотного луня в долине реки Барычи, Польша

Представлены результаты исследований популяции болотного луня проведенных в 1972–75 годах, когда еще применяли ДДТ и производился отстрел, и результаты исследований проведенных в 1982–84 годах, когда оба эти фактора не имели места на протяжении уже 7 лет. С конца XIX века численность популяции болотного луня возрастала в двух периодах (II мировая война и когда перестали применять ДДТ). Период регресса наблюдался во время наиболее интенсивного применения ДДТ (1950–1970). Плотность пар в разные периоды составляла 27–100 га/1 гнездовая пара. Площадь гнездового ареала в годы наиболее высокой численности составляла в среднем 160 га. У 4% самок наблюдалась бигамия. Их продукция птенцов была на 40% ниже чем у самок моногамных. У бигамных самцов продукция птенцов была на 30% выше чем у моногамных. 50% самок начивало кладки 27 апреля. Они сносили 2–8 яиц, в среднем 5.0. В начале сезона средняя кладка составляла 5.6 яиц, а под конец 4.4 яйца. Первые и последние яйца в кладке были как правило более мелкие по сравнению с остальными. Не констатировано различий в продукции летающих молодых в зависимости от величины яйца. Птенцы съедали в течение суток порцию пищи, масса которой была равна $^1/_4$–$^1/_3$ массы их тела. Самцы начинали летать по истечении 37–39 дней с момента выклева, самки по истечении 41–44 дней от выклева. Самцы, более легкие чем самки, были в состоянии схватить мало подвижную добычу на 54 дне жизни. Однако, родители докармловали молодых до 80 дня жизни. Соотношение полоз у молодых составляло 54% на пользу самцов. Выводки состояли обычно из птиц обоих полов, но 26% были однополыми. В кладках содержащих 5 яиц из более крупных яиц, откладываемых в средине периода кладки, обычно вылупливались самцы, из более мелких (начало и конец откладывания) – самки.

В период 1972–75 годов самые высокие потери были вызваны раздавливанием яиц высиживающими самками, а также тем, что из значительного количества яиц не выклевывались птенцы – и одно и другое было результатом воздействия на организмы болотных луней ДДТ. Индекс толщины скорлупы болотных луней из долины Барычи был на 23% ниже, чем у этого вида в Европе до 1945 года. Величины потерь по поводу ДДТ были настолько велики, что переменился нормальный у птиц тренд, согласно которому потери в начале гнездового периода являются меньшими по сравнению с потерями под конец периода. Длительные осадки увеличивали потери вызванные раздавливанием яиц и смертностью птенцов. В период 1982–84 причиной наибольших потерь было хищничество и оставление гнезд. Смертность молодых на гнездах составляла 25%. Птенцы, которые вылупились как первые, выживали до вылета из гнезда в 94%. Из птенцов, которые вылупились как шестое, не выжил ни один. Потери возникшие в результате агрессии в пределах выводка не имели значения. В период 1972–75 потенциальная продукция была реализирована в 42% а в период 1982–84 в 51%. Продукция молоди у пар с удачными выводками была одинакова в обоих периодах (3,1 молодой птицы на 1 пару). Причиной снижения численности болотного луня в период применения ДДТ было исключение некоторых пар из процесса размножения и сокращение длительности жизни взрослых особей вследствие отравления ДДТ и продуктами его обмена.

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В гнездовом периоде пища болотного луня в долине Барычи (N = 1355 жертв) состояла в 60% из птиц, 28% из преимущественно мелких млекопитающих, 10% из неживых рыб, а 2% составляли лягушки. 58% жертв добытых самцами происходило из сухопутных биотопов, а 24% из водных. У самок наоборот – 8% было сухопутных и 83% водных. Констатировано четкое разделение пищевых ниш самца и самки у болотного луня в гнездовом периоде. Жертвы самцов были в среднем на половину легче, чем жертвы самок.

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INTRODUCTION

In the recent 20-year period, birds of prey received a special attention because of a drastic decline in many species, including extinction of some populations.

This group of birds started to be threatened more than one hundred years ago. The highest level of threat was reached in the 1960s as a result of the appearance

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of a new, destructive factor such as different toxic substances introduced to ecosystems by man in the form of heavy metals and pesticides, especially DDT. The most affected were the species largely feeding on birds or fish (NEWTON 1979). However, the first signals concerning side effects of DDT leading to a reduction of breeding success due to shell thinning, and indicating also further consequences of this process appeared relatively late (RATCLIFFE 1967, PEACALL 1970).

The marsh harrier has a wide food spectrum (UTTENDÖRFER 1952, SCHIPPER 1973). In some populations, insectivorous passerines account for a large proportion of their diet (HILDEN & KALINAINEN 1966, BOCK 1979). In most European countries, the numbers of marsh harriers considerably declined in the period of DDT application (CRAMP & SIMMONS 1980). This was also the case in the Barycz valley in western Poland. Although at the end of the 1940s their numbers were low (SZARSKI 1950), a decline by 40% was noted in the 1960s, when the harmful effects of DDT were most pronounced.

The studies on marsh harriers in the Barycz valley were started in 1972, when destructive effects of DDT were already known, and many countries ceased using it. At the same time demands for protecting all species of birds of prey were growing.

In those years also the 4th volume of Handbuch der Vögel Mitteleuropas (GLUTZ *et al.* 1971) appeared in the European literature, summing up the known facts of the life of marsh harriers. This monumental work reveals many gaps and controversial issues concerning the biology of this species.

The study undertaken in 1972 on a sample involving then about 30 pairs was planned as a standard population study in the breeding season, but its most important purpose was to understand the factors accounting for the population decline in the study area. A special attention was paid to losses caused by the use of DDT, in particular to egg breakage by incubating females.

A relationship was expected between the food composition and the cumulation of DDT in body tissues. Thus, food samples were taken showing not only the diet composition but also the proportions of all prey categories, including those which cannot be detected in the remains left on the nest, or on feeding sites, or in pellets.

The methods used to estimate breeding failures caused by DDT (frequent visits to nests since the initiation of laying) and to analyze food composition on some occasions provided new details on breeding biology, including laying intervals, span of incubation period, division of parental care, food consumption in young of different age classes, growth rate of nestlings, daily activity of young and adult birds, sex ratio in the young generation, etc. Many of these aspects were not known from the literature on the marsh harrier.

Several years ago, it was noted that in some species eggs differ in size according to the sequence of their laying, and it was concluded that these differences are adaptive (HOWE 1976, SLAGSVOLD *et al.* 1984). Some authors found a relationship between egg size, the sequence of laying individual eggs, and the sex of

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nestlings hatched from these eggs (FIALA 1981, ANKNEY 1982). Using detailed materials collected for marsh harriers in 1972–75, this problem is also considered in the present paper. This was the period in which DDT was commonly used in Poland. This practice was officially forbidden in 1975, and since that time all birds of prey have been protected by law.

The same population was studied again in 1982–84, that is seven to nine years after the elimination of two important factors reducing population numbers – DDT and shooting. Generally, the study was limited to the breeding success and the causes of breeding failures. Supplementary food samples were also taken and some observations were made on the young after fledging. The main purpose of the study in the second period was to collect data permitting a comparison of reproductive parameters in the two periods.

STUDY AREA

The study was carried out in the middle reaches of the Barycz river (51°27′ – 51°35′ N, 16°58′ – 17°32′ E), where 6 complexes of fish-ponds are located. Going eastward these are: 1) Radziądz plus Niezgoda (830 ha), slightly to the south, 2) Jamnik (327 ha), 3) Ruda Sułowska (730 ha), 4) Stawno (1730 ha), 5) Potasznia (670 ha), and further to the south, 6) Wierzchowice (798 ha).

Complex 3, 4, and 5 are located in open areas. Complexes 1 and 2 border on a forest to the south, and almost all ponds of complex 6 are located in a forest, but no more than 1 km from open areas.

Soils are poor, sandy-podzolic. The main crop plants are potatoes, rye, and there are meadows and pastures there.

Human population in this part of the valley is low (49 inhabitants per 1 km^2). The largest town Milicz, has a population of 10 thousand people.

All the complexes, except complex 6, are in part nature reserves established in 1949 for the protection of birds.

A detailed study was conducted in complex 4. In the other complexes only numbers of breeding pairs of marsh harrier were estimated.

Complex Stawno (Fig. 1) is made up of 31 ponds of a joint surface area of 1730 ha. Suitable nesting sites occurred on 19 ponds, but marsh harriers nested on only 16 ponds, where the area of reed beds along the shores and on islands represented 14–50%, with a mean of 27% of the pond surface (with one exception). The other ponds, without nesting sites, were used as hunting grounds which offered plentiful nestlings of grebes, coots, black-headed gulls and wild ducks.

The ponds were surrounded by open areas. Cultivated land was slightly predominated by meadows. To the north and south the area was closed by forests growing on either side of the valley.

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No difference was found in the survival of the young between the early and late nests (Tab. 40). The mortality of the young in nests started in the first and in the second half of the season was about 25%. Thus, the significant differences in productivity between early and late nests were not due to differences in the mortality of the young.

NEST-SITE AND NATAL-SITE TENACITY

Adult birds. Several males with individual characteristic features returned to the same territories in successive years (one of them in four successive breeding seasons). This was also the case of one female. This nest site tenacity is also supported by the fact that the location of nesting territories and nest sites is similar or even identical from season to season in many cases.

Young birds. In the two study periods, all the young birds were ringed prior to fledging. If they showed a strong natal-site tenacity, it might be predicted that some of the 52 young ringed in 1972 would breed in the study area in 1974 and all of them in 1975.

In 1974, 1 male and 2 females were ringed among the breeding birds, and in 1975, 4 females. In 1982–84, the ringed breeding birds included 2 males and 4 females in 1982, 2 males and 6 females in 1983, and 1 male and 5 females in 1984. It is possible that the number of ringed males was underestimated, but the number of ringed females certainly was exact.

These data suggest that the natal-site tenacity in young birds is low, provided their mortality was not extremely high in the period between fledging and the beginning of reproduction.

FOOD

Over the two study periods, 1355 prey items were identified (Tab. 41). The material is divided into two groups:

a) data from whole-day observations at nests from the hide,

b) remains picked up from nests and feeding platforms, and data from observations of hunting marsh harriers.

Prey remains were collected from platforms, and observations were made since return of marsh harriers from their winter quarters. Pellets were not collected.

The material of the first group is of the greatest value. The observations were made for 85 days at different nests. They provide information on the food spectrum over the breeding season, and also show actual proportions of different items in the diet, which cannot be obtained by any other method since some prey (*e.g.* naked passerine nestlings and young of small mammals) are absent from the remains on the nest or in pellets.

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Table 41. Food composition of marsh harriers in the breeding season in the Barycz valley
Tabela 41. Skład pokarmu błotniaków stawowych w okresie lęgowym w dolinie Baryczy

Prey Zdobycz	Prey recorded by nest observations from a hide Zdobycz stwierdzona podczas obserwacji gniazd z ukrycia		Prey recorded by field observations and nest and platform checking Zdobycz stwierdzona podczas obserwacji w terenie oraz kontroli gniazd i platform		Total Razem	
	Number Liczba	%	Number Liczba	%	Number Liczba	%
1	2	3	4	5	6	7
Microtus arvalis	156	19.1	39	7.1	195	14.4
Microtus sp.	14	1.7	3	0.5	17	1.3
Arvicola terrestris	12	1.5	19	3.5	31	2.3
Lepus europaeus	12	|	11	2.0	23	1.7
Ondatra zibethicus	10	1.2	10	1.8	20	1.5
Murinae unidentified	10	|	19	3.5	29	2.1
Apodemus agrarius	3	0.4	1	0.2	4	0.3
Mus musculus	3	|	–	–	3	0.2
Rattus norvegicus	3	|	2	0.4	5	0.4
Talpa europaea	3	|	9	1.6	12	0.9
Clethrionomys glareolus	1	0.1	–	–	1	0.1
Sorex araneus	1	|	10	1.8	11	0.8
Crocidura sp.	–	–	2	0.4	2	0.1
Erinaceus europaeus	–	–	1	0.2	1	0.1
Sus scrofa (carrion)	–	–	2	0.4	2	0.1
Total Mammals – Razem Ssaki	228	27.9	128	23.3	356	26.3
Fulica atra	74	9.1	94	17.1	168	12.4
Alauda arvensis	69	8.4	18	3.3	87	6.4
Acrocephalus scirpaceus	63	7.7	2	0.4	65	4.8
Aythya ferina	38	4.7	17	3.1	55	4.1
Anatidae unidentified	26	3.2	3	0.5	29	2.1
Aythya fuligula	21	2.6	10	1.8	31	2.3
Podiceps cristatus	19	2.3	23	4.2	42	3.1
Sturnus vulgaris	18	2.2	23	|	41	3.0
Emberiza schoeniclus	18	|	13	2.4	31	2.3
Podiceps sp.	15	1.8	14	2.6	29	2.1
Acrocephalus arundinaceus	7	0.9	1	0.2	8	0.6
Rallus aquaticus	6	0.7	6	1.1	12	0.9
Perdix perdix	6	|	3	0.5	9	0.7

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1	2	3	4	5	6	7
Passer montanus	6	6.7	2	0.4	8	0.6
Emberiza citrinella	6	|	1	0.2	7	0.5
Passer sp.	5	0.6	2	0.4	7	|
Saxicola rubetra	5	|	–	–	5	0.4
Passer domesticus	4	0.5	5	0.9	9	0.7
Podiceps nigricollis	2	0.2	6	1.1	8	0.6
Podiceps ruficollis	2	|	5	0.9	7	0.5
Aythya nyroca	2	|	1	0.2	3	0.2
Larus ridibundus	2	|	18	3.3	20	1.5
Cuculus canorus	2	|	3	0.5	5	0.4
Gallinula chloropus	2	|	1	0.2	3	0.2
Turdus pilaris	2	|	3	0.5	5	0.4
Turdus merula	2	|	2	0.4	4	0.3
Turdus philomelos	2	|	–	–	2	0.1
Phylloscopus sp.	2	|	–	–	2	0.1
Vanellus vanellus	1	0.1	3	0.5	4	0.3
Sterna hirundo	1	|	2	0.4	3	0.2
Porzana parva	1	|	1	0.2	2	0.1
Carduelis carduelis	1	|	1	|	2	|
Columba domestica	–	–	1	|	1	|
Chlidonias niger	1	0.1	–	–	1	|
Dendrocopos major	1	|	–	–	1	|
Lanius collurio	1	|	–	–	1	|
Coccothraustes coccothraustes	1	|	–	–	1	|
Carduelis chloris	1	|	–	–	1	|
Emberiza calandra	1	|	–	–	1	|
Passeriformes unidentified	53	6.5	20	3.6	73	5.4
Anser anser	–	–	19	3.5	19	1.4
Total Birds – Razem Ptaki	489	59.9	323	58.8	812	59.9
Natrix natrix	–	–	2	0.4	2	0.1
Lacerta sp.	–	–	1	0.2	1	0.1
Total Reptiles – Razem Gady	–	–	3	0.5	3	0.2
Rana sp. (*esculenta*)	18	2.2	12	2.2	30	2.2
Rana sp. (*temporaria/arvalis*)	1	0.1	10	1.8	11	0.8
Total Amphibians – Razem Płazy	19	2.3	22	4.0	41	3.0
Cyprinus carpio	81	9.9	61	11.1	142	10.5
Abramis brama	–	–	1	0.2	1	0.1
Total Fishes – Razem Ryby	81	9.9	62	11.3	143	10.6
Waterfowl's eggs Jaja ptaków wodnych	–	–	numerous liczne		numerous liczne	

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The main food items of marsh harriers in the Barycz valley consisted of birds (60%), mammals (28%), fish (10%), and frogs (2%). *Passeriformes* accounted for 55% of the avian prey, and small mammals were dominated by rodents (82%). Fishes, with one exception, were represented by carps, mostly freshly dead, two-year-old individuals.

Diet of marsh harriers from their return in spring to the beginning of incubation (20 March–30 April)

In this period, the diet of marsh harriers can only qualitatively be characterized. This was the most difficult period to them in terms of prey abundance and availability. The remains of food on platforms and the prey caught by hunting individuals consisted mostly of starlings *Sturnus vulgaris*, frogs, fish, and eggs of waterfowl nesting early in the season. At that time, as in later periods, small mammals must have accounted for much of the diet, but their remains were scanty. Starlings were hunted on the land and also in the reedbed where they roosted at night. Each day marsh harriers attentively searched such roosting sites. Apparently many weak or dead starlings remained in them. This was also the mating period of frogs *Rana temporaria* and *Rana arvalis* and they aggregated in less dense reeds growing in shallow water.

The early April was the beginning of laying of coots and mallards. A little later also grebes, occurring in large numbers on the ponds, started laying. About mid-April, grey-lag geese hatched.

In this period, eggs of coots and mallards *Anas platyrhynchos* were important components of the diet. I cannot determine their percentage in the diet, but judging from the number of nests of these two species robbed by marsh harriers, it must have been significant. Eggs were eaten on the nest of the prey or near by. Only occasionally were they carried to platforms.

Grey-lag goslings probably did not represent a significant proportion of the marsh harrier diet, although in the first study period 132–154 pairs of this species nested on the ponds of complex Stawno, and in the second period about 180 pairs (author's own data). Goslings were well defended by their parents. Nevertheless, I observed a marsh harrier eating a gosling at the edge of the nest in the presence of an old goose brooding the remaining young.

Diet in the nesting period

The 62 prey items delivered to the incubating females by their mates consisted of small mammals, mostly common voles *Microtus arvalis* (63%), birds, mostly skylarks *Alauda arvensis* (29%), also frogs and fish (8%), but only on rainy days.

The food delivered to the small young did not differ from the food delivered to

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the incubating females. Among small birds, many nestlings and fledglings were caught. Later on, harriers preyed upon chicks of larger water birds (coots, mallards, grebes) and young hares. As long as the food was delivered only by the male, the bulk of the diet consisted of terrestrial animals which were relatively small.

As the young grew older and the female started hunting, the number of prey from aquatic habitats increased. In the order of their decreasing proportion in the diet these were nestlings of grebes, ducks, coots and other *Rallidae*. Among mammals there were young muskrats *Ondatra zibethicus* and vole-rats *Arvicola terrestris*. Also the number of dead carps increased in the diet. Among small prey, nestlings of reed warbler were numerous.

The period when the young marsh harriers had highest food requirements was well correlated with the greatest abundance of food, especially in aquatic habitats. At that time the young of most water birds abundant on the ponds and representing the potential prey of marsh harriers, were already reared (WITKOWSKI 1967, DYRCZ 1981, JANKOWSKI 1983).

Diet in the post-fledging period (15 August–30 September)

Sources of information concerning the diet in this period were scarce, based mainly on observations of hunting sites and estimates of potential prey in them.

In this period, not fully grown ducks, coots and grebes were rare. Young passerine birds were flying equally well as adults, and started migratory movements. In a word, the importance of aquatic habitats as the source of food for marsh harriers sharply declined, especially as compared with the preceding period. The remains on feeding platforms comprised fish, frogs, scarce coots, tufted ducks *Aythya fuligula*, and delayed black-headed gulls.

Many marsh harriers left the ponds and started nomadic movements. They hunted more often in farmland than on ponds. Harvest was just over, and small mammals, especially common voles, were then more available than any other prey on the ponds. But as soil fertility was low in the region of Stawno, no wonder that marsh harriers gave up hunting on crop fields very soon. They probably moved to the adjacent Oder valley, where crop fields were much more fertile and abounded in small rodents. For example, they were frequently observed between Trzebnica and Wrocław, where they certainly did not breed.

Differential food niches of males and females

A marked change in the food delivered to the young when the females started hunting suggested that the two sexes had different food niches. As the females delivered to the nest not only their own prey but also those passed by the male, it

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was not possible to know which bird hunted a given prey when observing the nest from a hide in the reed. This problem was solved in two ways (see p. 232–233). Four nests were observed in different years to separate the prey hunted by each parent. The total material consisted of 441 prey items of which 252 were hunted by the male (Tab. 42) and 189 by the female (Tab. 43).

For all the nests the prey delivered by the male consisted mainly of animals from open land habitats (Tab. 44). They accounted for 70% of all prey items in two males (A and C). Only for one male (B) was this proportion below 50%; nevertheless land prey outweighed the prey from water habitats.

Table 42. Male's prey estimated in four pairs (A, B, C, D) of marsh harriers in the Barycz valley
L – prey from land habitats, W – from water habitats, N – from indetermined habitats

Tabela 42. Zdobycze samców z 4 różnych par (A, B, C, D) błotniaków stawowych w dolinie Baryczy
L – zdobycz pochodząca z siedlisk lądowych, W – siedlisk wodnych, N – siedlisk nie ustalonych

Prey Zdobycz	Habitat Siedlisko	Male – Samiec				Total Razem	Mean weight (g) Średnia masa (g)
		A	B	C	D		
Microtus arvalis	L	34	27	13	11	85	25
Alauda arvensis	L	18	9	4	4	35	30
Passeriformes unidentified	N	10	16	–	3	29	15
Fulica atra (pulli)	W	1	9	3	2	15	125
Podiceps cristatus (pulli)	W	3	5	1	1	10	100
Lepus europaeus (juv.)	L	6	2	–	–	8	200
Microtus oeconomus/agrestis	N	3	3	1	–	7	40
Podiceps sp. (pulli)	W	–	5	1	1	7	70
Murinae unidentified	L	–	5	1	–	6	20
Anatidae unidentified	W	–	5	1	–	6	60
Aythya fuligula (pulli)	W	3	2	–	–	5	60
Sturnus vulgaris	N	1	4	–	–	5	70
Emberiza schoeniclus	N	1	3	–	1	5	15
Cyprinus carpio	W	2	2	–	1	5	150
Aythya ferina (pulli)	W	–	4	–	–	4	60
Perdix perdix (pulli)	L	3	–	1	–	4	50
Talpa europaea	L	2	1	–	–	3	100
Acrocephalus scirpaceus	W	3	–	–	–	3	12
Passer montanus	L	3	–	–	–	3	25
Rana sp. (*esculenta*)	W	–	1	1	1	3	20
Ondatra zibethicus (juv.)	W	1	–	–	–	1	200
Arvicola terrestris (juv.)	W	1	–	–	–	1	70
Mus musculus	L	1	–	–	–	1	20
Clethrionomys glareolus	L	1	–	–	–	1	20
Total – Razem		97	103	27	25	252	46.8

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Table 43. Female's prey estimated in four pairs (A, B, C, D) of marsh harriers in the Barycz valley
L – prey from land habitats, W – from water habitats, N – from indetermined habitat

Tabela 43. Zdobycze samic z czterech par (A, B, C, D) błotniaków stawowych w dolinie Baryczy
L – zdobycz pochodząca z siedliska lądowego, W – z siedliska wodnego, N – z siedliska nie ustalonego

Prey Zdobycz	Habitat Siedlisko	Female – Samica				Total Razem	Mean weight (g) Średnia masa (g)
		A	B	C	D		
Cyprinus carpio	W	20	2	11	4	37	150
Fulica atra (pulli)	W	14	5	3	8	30	130
Acrocephalus scirpaceus (pulli)	W	11	4	3	3	21	12
Rana sp. (*esculenta*)	W	1	5	1	5	12	20
Passeriformes unidentified	N	5	2	2	2	11	15
Aythya ferina (pulli)	W	1	4	2	2	9	90
Anatidae unidentified	W	4	–	4	1	9	90
Microtus arvalis	L	2	3	2	1	8	25
Podiceps sp. (pulli)	W	4	3	1	–	8	120
Arvicola terrestris	W	5	–	1	1	7	80
Aythya fuligula	W	5	–	2	–	7	90
Ondatra zibethicus (juv.)	W	2	–	1	1	4	200
Podiceps cristatus	W	2	–	2	–	4	200
Rallus aquaticus	W	3	1	–	–	4	40
Passer sp.	L	2	–	1	1	4	25
Cuculus canorus (pulli)	N	2	–	1	–	3	60
Alauda arvensis	L	1	1	–	1	3	25
Sturnus vulgaris	N	–	1	1	–	2	70
Acrocephalus arundinaceus	W	2	–	–	–	2	25
Emberiza schoeniclus	N	1	1	–	–	2	15
Podiceps ruficollis (pullus)	W	–	1	–	–	1	200
Larus ridibundus (pullus)	W	1	–	–	–	1	300
Total – Razem		88	33	38	30	189	88.6

Not less than 70% of the prey caught by females represented aquatic animals. This provides evidence that male and female of marsh harriers of the Barycz valley had clearly separated food niches. Males typically hunted in meadows and crop fields, and females on ponds.

The prey hunted by females was twice as heavy as the prey hunted by males (Tabs 42 and 43).

The diet of marsh harriers did not include birds larger than fully grown moorhen *Gallinula chloropus* and mammals larger than a big rat. All prey items bigger than these two species and occurring in the diet of Barycz valley marsh harriers (Tabs 41–43) were young individuals, at the most half grown.

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Population estimates of birds in Great Britain and the United Kingdom

Ian Woodward, Nicholas Aebischer, Daisy Burnell, Mark Eaton, Teresa Frost, Colette Hall, David Stroud and David Noble



Dan Powell

Northern Lapwing *Vanellus vanellus*

Abstract Population estimates of birds have many applications in conservation and ecological research, as well as being of significant public interest. This is the fourth report by the Avian Population Estimates Panel, following those in 1997, 2006 and 2013, presenting population estimates of birds in Great Britain and the United Kingdom. Overall, there are thought to be about 84 million breeding pairs of birds in the UK, similar to the total in *APEP 3*. The Wren *Troglodytes troglodytes* continues to be the most common species, with a current estimate of 11 million pairs, which has increased since *APEP 3*. There are more than one million pairs of 20 species (23 in *APEP 3*) and these 20 species contribute 77% of the total (58% of the total is accounted for by the ten commonest species alone). Population estimates originate from a wide variety of sources, many involving extrapolation of previous estimates by recognised trend measures. Despite the often exceptionally detailed information available on bird numbers, there remain many gaps in our knowledge. Recommendations are made to improve our understanding of bird numbers nationally.

Introduction

Knowing the absolute number of birds in a population is important to those who make decisions about conservation policy and engage in site conservation management, but is also of public interest. However, it can be difficult to produce robust estimates of bird population size. First, numbers fluctuate from year to year and even from month to month as individuals breed, die and migrate (Newton 1998). Second, for all but the scarcest species, it is usually impossible to carry out a full census (i.e. count every individual), and hence population estimates are based on surveys that provide information for a sample of sites and use statistical techniques and assumptions about behaviour, detectability and representativeness in order to turn counts into population estimates. This raises inevitable uncertainties about the accuracy of population estimates (Hewson *et al.* 2018).

Although trends over time are particularly valuable for assessing the status of species and biodiversity for many conservation purposes, knowledge of the absolute size of an animal population is also needed to fully understand threats to that species, to evaluate the risk of extinction and to make decisions about how to protect it. One of the most important uses of national and international population estimates in conservation is to enable sites to be designated and hence protected on the basis that they support an important part of a species' population, usually 1% of the international or national population (Ramsar Convention 2012; Drewitt *et al.* 2015; IUCN 2016). Absolute population estimates are also important in assessing extinction risk: species with a small population are more likely to be at risk of extinction, particularly if a large proportion of that population is found at a small number of sites.

Population size estimates are therefore a key conservation tool alongside population trends and other information. The European Union (EU) Directive on the conservation of wild birds (2009/147/EC) requires Member States to report on the status of native bird species every six years. This report under Article 12 includes an assessment of species population status (population sizes and distributions, and changes in these parameters over time). EU reporting under Article 12 was added to information collated in parallel from non-EU countries in order to update the European Birds of Conservation Concern (Staneva & Burfield 2017), which followed



David Tipling

26. The UK's breeding population of Turtle Doves *Streptopelia turtur* is declining rapidly. The first APEP report (in 1997) put the population at 75,000 pairs, which compares with the current estimate of just 3,600 pairs in 2016 (and even that may now be too high); UK, April 2009.



Mark Lewis

27. Arctic Skua *Stercorarius parasiticus*, Sanday, Orkney, May 2017. The population estimate for Arctic Skua, and seven other seabirds, has been extrapolated from trends since the last complete national survey, *Seabird 2000*, which ran from 1998 to 2002. The current national census, *Seabirds Count*, is expected to be complete in 2020 and new counts for all species should be available soon.

on from previous Birds in Europe publications (e.g. BirdLife International 2004).

The Avian Population Estimates Panel (APEP) is a collaboration between the UK statutory conservation agencies and relevant non-governmental organisations. The role of APEP is to collate the best estimates of breeding and non-breeding bird population size and present a consensus view on the most appropriate estimates for relevant conservation applications, such as defining thresholds for statutory site designations. The APEP process was endorsed in 2002 by the Joint Nature Conservation Committee (JNCC), which acknowledged the value to the conservation agencies of a single, quality-assured source of population estimates for statutory conservation purposes (Stroud *et al.* 2002). Three previous APEP assessments have been published in *British Birds*: *APEP 1* (Stone *et al.* 1997), *APEP 2* (Baker *et al.* 2006) and *APEP 3* (Musgrove *et al.* 2013).

This report (*APEP 4*) presents the most recent estimates for both Great Britain (GB) and the United Kingdom (UK). Most of these estimates were submitted, together with other data and information, as part of the UK's Article 12 report to the EU in September 2019 (JNCC 2019).

Methods

Taxonomic scope: species, subspecies and biogeographic populations

All species in Categories A and C of the British List (BOU 2017) were considered for inclusion in the main species table (table 1); other non-native species were not listed in previous APEP reports but, for completeness, all non-natives for which breeding has been confirmed since 2011 are listed here in table 3. Population estimates were sought mainly at species level, but for 18 species it was possible to give estimates for some subspecies or geographical populations (table 2). This is of particular significance where the subspecies has a distinct legal status or is endemic. It was not possible to give estimates for three currently recognised endemic subspecies owing to a lack of knowledge about their distribution: Meadow Pipit *Anthus pratensis whistleri*, Linnet *Linaria cannabina autochthona* and Yellowhammer *Emberiza citrinella caliginosa* (Parkin & Knox 2010).

In addition, for some waterbirds it is well-established practice to base conservation policy on separately delineated biogeographic populations; we follow the treatment used by Frost *et al.* (2019b) in this regard.

Population estimates are presented for 249



Andrew Bloomfield

28. Breeding Eurasian Spoonbills *Platalea leucorodia* (with juvenile Grey Herons *Ardea cinerea* to the left), Holkham, Norfolk, May 2011. The Spoonbill is one of several wetland bird species presently colonising the UK.

species during the breeding season, 113 species during winter and for one species during passage. The estimates for 18 species are broken down into subspecies for either the breeding season or winter (or both), and are presented for a total of 29 breeding and 12 wintering subspecies.

Seasonal scope: breeding, wintering and passage populations

Most estimates are for the breeding season. Breeding estimates are presented for all species included in *APEP 3* and for additional species (including non-natives) with at least one case of proven breeding from 2011 onwards.

The precise definition of the non-breeding season varies according to species and estimation technique (as set out in this paper and in sources referenced). In general, non-breeding estimates have been omitted for largely resident species, except for waterbirds where statutory site protection and reporting is based around non-breeding estimates; this repeats the approach used for *APEP 3*. Apart from waterbirds, winter estimates have been produced only for the small number of wintering species included in *APEP 3*. For some other species (such as Common Chaffinch *Fringilla coelebs* and Blackbird *Turdus merula*), estimates of non-breeding numbers have again not been attempted owing to lack of suitable data sources, even though it is known that resident populations are supplemented in winter by large-scale arrivals.

Estimates of passage numbers have been excluded, with the exception of the globally threatened Aquatic Warbler *Acrocephalus paludicola*.

Geographic scope

As far as possible, separate estimates have been produced for the geographical entities of Great Britain and the UK (although the two estimates are sometimes identical following rounding; see below). All estimates exclude the Channel Islands. The Isle of Man (IoM) is not part of GB nor the UK but given the available data sources and methods, its exclusion from most estimates is generally not straightforward, and thus most estimates are actually those for GB+IoM and UK+IoM. For most species this makes little difference but in cases where the IoM contributes a significant and calculable proportion of the GB or UK total, separate IoM estimates are presented (and in those cases, strict GB and UK estimates are also given): IoM estimates are given in table 4 for Ringed Plover *Charadrius hiaticula*, Herring Gull *Larus argentatus*, Little Tern *Sternula albifrons*, Hen Harrier

Circus cyaneus, Peregrine Falcon *Falco peregrinus* and Red-billed Chough *Pyrrhocorax pyrrhocorax*.

For many conservation purposes, an estimate for the UK – i.e. GB plus Northern Ireland (NI) – is required. However, for Special Protection Area (SPA) and Sites of Special Scientific Interest (SSSI) selection, the geographical framework is either GB or All-Ireland (in the case of site Areas of Special Scientific Interest selection in Northern Ireland). If separate estimates for GB and the UK were available from the original source, these are presented. However, many original estimates, in particular those in *Breeding Atlas 1988–91* (Gibbons *et al.* 1993), were for GB or for All-Ireland. In some cases, such as many wintering waterbirds, estimates from NI were also available, and were added to the GB estimate to produce a UK estimate. Where the original source estimate was a UK figure, such as in the Rare Breeding Birds Panel report (RBBP; Holling *et al.* 2019), a NI estimate has been subtracted to produce the GB estimate. If NI-specific estimates were not available, then the UK estimates have been derived from extrapolated GB estimates using Atlas data, as described below.

Offshore wintering populations

Frost *et al.* (2019b) incorporated aerial and boat-based survey data from British waters into the wintering estimates for some marine species. British marine waters extend up to 200 nautical miles (370.4 km) or to an international boundary, if closer. In practice, there have been relatively few such surveys and many are commercial surveys from which data are not in the public domain. Consequently, most estimates exclude offshore areas out of sight of land. For 17 species that were suspected or known to have significant offshore distributions, the figures presented are marked as known underestimates in table 1 and include mostly birds in the near-shore environment only. Limited offshore data were included in previous estimates for Common Scoter *Melanitta nigra* and Red-throated Diver *Gavia stellata*, but more aerial survey data have become available subsequently for these species (Frost *et al.* 2019b). Although some of these data were used, the totals are still considered an underestimate for British waters.

Timescale and extrapolation

For each combination of species, geographical area and season, the Panel sought the most recent published estimate, along with its confidence limits (if any), units, date (or date range), method of derivation and reference. The aim has been to collate the most recently published estimates. Typically, these relate approximately to the period 2013–17 (for breeding estimates) or 2012/13–2016/17 (for winter). However, as in previous APEP



Richard Chandler

29. The Wren *Troglodytes troglodytes* remains the species with the highest population estimate in the UK, with a new estimate of 11 million pairs; Norfolk, May 2015.



Richard Chandler

30. Oystercatchers *Haematopus ostralegus* piping at their territorial boundary (left two birds are one pair, the right two birds are the other pair; far left and far right are males, centre birds both females); Snettisham, Norfolk, March 2012. The Oystercatcher and Northern Lapwing *Vanellus vanellus* are two declining species of breeding wader for which population estimates have dropped below 100,000 pairs for the first time in this report.

reports, many of the breeding estimates are based on extrapolation of previously published estimates using breeding trend data. In these cases, breeding-season estimates have generally been extrapolated from given baselines to 2016 (as described below). In addition, for older estimates (from 2012 or earlier) the possibility of extrapolating estimates forward in time was investigated. In some cases, more than one estimate was available for a given species/region/season combination; as in previous APEP reports, the Panel discussed the reasons behind such discrepancies and came to a collective view on the most appropriate estimate.

The methodology for extrapolating from estimates from 2012 or earlier was described in detail in *APEP 3* and is not repeated fully here. For most species for which this method of extrapolation was used, the UK estimate was derived from a GB estimate using data from *Bird Atlas 2007–11* (Balmer *et al.* 2013) in the same way as in *APEP 3*. The extrapolation used was a combination of two factors. First, the ratio of numbers of 10-km squares with possible, probable or confirmed breeding evidence in GB and UK was calculated. Second, a measure of average density in both GB and Ireland was derived by comparing results of *Bird Atlas 2007–11* timed tetrad visits (first hour only, and adjusted to account for differing levels of coverage in different areas).

For a small number of species for which the UK estimate was derived from the GB estimate in *APEP 3*, a more recent NI estimate was available for *APEP 4*, either from a species survey in NI or from the Irish Rare Breeding Birds Panel (IRBBP) reports covering 2012–16 (Perry 2013; Perry & Newton 2014; Newton 2015, 2016). For these species, the GB estimate is based on extrapolation from *Breeding Atlas 1988–91* (as before), and the NI estimate has been added to produce a UK figure. In the case of Northern Lapwing *Vanellus vanellus*, Eurasian Curlew *Numenius arquata* and Common Snipe *Gallinago gallinago*, the UK estimates in table 1 include figures from the breeding wader survey in 2013 (Colhoun *et al.* 2015). For other breeding species for which a NI estimate was available, the NI population was negligible. The source for each NI estimate (usually IRBBP) is listed in the table wherever it has been used to produce raw (unrounded) UK estimates, even if the rounded GB and UK estimates remain the same.

As in *APEP 3*, extrapolations use unrounded estimates whenever they are available in order to avoid the risk of unnecessary multiple rounding (i.e. taking a rounded estimate, extrapolating by a rounded trend, and then rounding the final answer again). In addition, all estimates presented here have been recalculated from original source estimates, rather than extrapolating from 2009

estimates presented in *APEP 3*. This is to avoid introducing potential errors from subsequent changes to the 2009 smoothed trend, caused by inclusion of additional BBS data that were not available when the *APEP 3* estimates were calculated.

Data sources and derivation methods

For most species, the data sources and derivation methods are the same as those used in *APEP 3*; see Musgrove *et al.* (2013). In a small number of cases they have changed, usually because data from a new single-species survey or new analyses of existing data have become available.

Summary details of each data source and derivation method are provided below to enable interpretation of the estimates presented in the tables and to highlight minor differences from the previous methods where appropriate. The numbers for each section are the same as those used to reference the derivation method in the tables.

1. Common breeding species: extrapolation from *Breeding Atlas 1988–91*

For many common species, the original GB estimates from 1988–91 have been extrapolated forward using exactly the same methodology as described above, and using the same trend measures. The extrapolation uses the most appropriate trend information available, usually a joint Common Birds Census (CBC)/Breeding Bird Survey (BBS) trend (Woodward *et al.* 2018) for estimates made prior to 1995, or a smoothed BBS-only trend (Harris *et al.* 2019) for estimates made since 1995 (BBS started in 1994). For a few species favouring waterways, the extrapolation uses a smoothed UK joint trend calculated from the Waterways Bird Survey (WBS, running from 1974 to 2007; Woodward *et al.* 2018) combined with the Waterways Breeding Bird Survey (WBBS, from 1998 onwards; Harris *et al.* 2019). Although *Bird Atlas 2007–11* has been published since *APEP 3*, new population estimates were not produced for that work and the 1988–91 estimates remain the most recent GB original source reference for many species. As UK estimates are derived from the extrapolated 2016 GB figure, the adjustments make use of the more concurrent *Bird Atlas 2007–11* timed count data, which present a robust measure of relative abundance between GB and NI.

As previously, there are a small number of species for which a robust CBC/BBS (or equivalent WBS/WBBS) trend could not be calculated. Extrapolated estimates for these species use the BBS trend from 1995 and hence make the assumption that there was no population change between 1988–91 and 1995. An exception is Red Grouse *Lagopus lagopus*, for which the estimate from *Breeding Atlas 1988–91* was extrapolated to 1995 by means of the Game & Wildlife Conservation Trust's National Gamebag Census trend (Aebischer 2019), recalibrated to density as per *APEP 2*, and then from 1995 to 2016 by the BBS trend. For one species, Rock Pipit *Anthus petrosus*, the estimate given in *Breeding Atlas 1988–91* has again been repeated as no suitable trend was identified to extrapolate it to 2016.

2. Distance-sampled BBS density-based estimates

For 24 common species, the population estimates in *APEP 3* were extrapolated from those produced by Newson *et al.* (2008), which used 2006 data from distance bands in BBS to infer the proportion of birds undetected by observers, and hence estimate how many birds were present in total. The method is computationally intensive so has not been repeated subsequently; and the estimates for 2006 have been extrapolated to 2016 using the BBS smoothed trend. As discussed fully in *APEP 3*, BBS density-based estimates work better for some species than others, depending on (for example) the relative detectability of males and females and the number of non-breeding birds present. The Panel carefully considered estimates from Newson *et al.* (2008) for *APEP 3* on a case-by-case basis, and used only those where there was clear evidence that the distance-based estimate was likely to be more accurate. This report uses this method for the same 24 species as *APEP 3*.

3. Rare breeding birds

The annual RBBP report publishes a population estimate for many scarce breeders to help monitor breeding trends, based on the mean

maximum reported population size over the most recent period for which data are available. For most species, the population estimates given here are the five-year means from the 2017 RBBP report (Holling *et al.* 2019), covering the period 2013–17; these are adjusted to exclude breeding records from the Channel Islands and IoM and also exclude mixed pairs. A five-year mean is preferred to data for a single year as it accounts for variation in reporting and detection. The population estimates presented here are thus equivalent to the maximum (higher) figure where a range was reported for rare breeding birds in *APEP 3* (minimum and maximum five-year mean estimates). Five-year means for non-native breeders have been calculated for the most recent five-year period for which data were available (2010–14). Where the mean maximum figure would round down to zero pairs, a range of 0–1 pairs is given.

For five species (Eurasian Spoonbill *Platalea leucorodia,* Eurasian Bittern *Botaurus stellaris,* Cattle Egret *Bubulcus ibis*, Great White Egret *Ardea alba* and White-tailed Eagle *Haliaeetus albicilla*), it was considered that recent increases seem likely to be sustained, and consequently that data from the most recent year is a more realistic assessment of the population than a five-year mean. Similarly, for the non-native Lady Amherst's Pheasant *Chrysolophus amherstiae,* the most recent information (Holling *et al.* 2018) suggests that this species is now extinct as a UK breeding species.

In isolated cases, older data are considered more reliable than the most recent figures. The *APEP 3* estimate for Common Goldeneye *Bucephala clangula*, covering the period 2006–10, is repeated here as there has been a reduction in monitoring intensity since 2011.

The estimates for Gadwall *Mareca strepera* and Red Kite *Milvus milvus* were based on RBBP data in *APEP 3* but both have since been dropped as RBBP species as they have become more abundant. The GB estimates in *APEP 3* for these species have been updated by extrapolation using BBS trends as described above. Both species are still covered by the IRBBP, and the NI mean total for the most recent five-year period was added to GB estimates to produce UK totals. Cetti's Warbler *Cettia cetti* was dropped from the RBBP list after 2016 and hence the 2016 RBBP estimate is used.



Edmund Fellowes/BTO

31. The Black Grouse *Lyrurus tetrix* is an example of a species for which a bespoke estimate was produced specifically for *APEP 4*; UK, April 2014.



Richard Chandler

32. Wintering populations of wildfowl, such as these Tufted Ducks *Aythya fuligula* in Northamptonshire in December 2011, are relatively well monitored by the Wetland Bird Survey (WeBS). More broadly, the populations of many other wintering species remain poorly understood.

4. Breeding seabirds

The estimates of numbers of breeding seabirds were based on *Seabird 2000*, carried out between 1998 and 2002 (Mitchell *et al.* 2004). The next national census (*Seabirds Count 2015–2020*) is not yet complete and hence the estimates for some species are the same as in *APEP 3*. These include Herring and Lesser Black-backed Gulls *Larus fuscus*, which increasingly nest in urban areas and for which traditional census techniques have become difficult.

Annual monitoring of some seabird colonies takes place under the Seabird Monitoring Programme (SMP) (https://jncc.gov.uk/our-work/seabird-monitoring-programme/) and thus some *Seabird 2000* population estimates have been extrapolated to 2015 using SMP trends. These were calculated using the long-term Thomas trend index (Thomas 1993; JNCC 2016) from 1986–2015 (using counts from a sample of colonies monitored by SMP), anchored to the *Seabird 2000* census count. Owing to concerns about how representative the colonies monitored by SMP are, updated estimates are presented here for only those eight species for which trends are considered a robust measure for the whole population. To test the robustness of the Thomas trend, an estimation of the *Seabird 2000* population was produced using the trend index between 1986 and 2000 and anchored to the 1986 Seabird Colony Register (SCR) census population for the species. If this estimate had a variance of 30% (or more) from the actual *Seabird 2000* population, the method was not used and the *Seabird 2000* census figure is repeated instead. All seabird estimates in table 1 are strict GB or UK estimates and exclude the IoM.

For four species, alternative estimates are used: Northern Gannet *Morus bassanus*, for which a full survey was carried out in 2013–14 (JNCC 2016); and Mediterranean Gull *Ichthyaetus melanocephalus*, Little Tern and Roseate Tern *Sterna dougallii*, for which annual data of sufficient quality are available via the RBBP.

5. Breeding estimates based on single-species surveys and bespoke analyses

As in previous APEP reports, breeding estimates have been sourced from the literature and include single-species surveys carried out under the SCARABBS programme (Statutory Conservation Agency and RSPB Annual Breeding Bird Scheme), as well as other dedicated surveys. Where possible, older published estimates have been extrapolated to 2016 as described above. For species where no suitable trend was available for extrapolation, the estimate relates to an earlier date, which is stated in the tables. For Grey Heron *Ardea cinerea*, estimates of the breeding population are derived annually from the BTO's long-running Heronries Census.

For five resident wildfowl species – Greylag Goose *Anser anser*, Canada Goose *Branta canadensis*, naturalised Barnacle Goose *B. leucopsis*, Egyptian Goose *Alopochen aegyptiaca* and Mandarin Duck *Aix galericulata*



Amy Lewis/BTO

33. The Red Kite *Milvus milvus* is an example of a former RBBP (Rare Breeding Birds Panel) species that is now too numerous to be covered by RBBP but which can now be monitored by BBS (Breeding Bird Survey), which has enabled the last RBBP population estimate to be updated; UK, June 2011.

– breeding population estimates were derived in the same way as for *APEP 3*, by dividing the most recent winter estimate by a standard value of three (Meininger *et al.* 1995).

For a small number of other species, novel analyses or compilations were undertaken for *APEP 4;* these are described in the footnotes in table 1. Since more recent data were not available for Tawny *Strix aluco*, Long-eared *Asio otus* and Short-eared Owls *A. flammeus*, the *APEP 3* estimates have been repeated for these species. The GB breeding estimate for Common Eider *Somateria mollissima* has been updated using the same method used for *APEP 3*; and new bespoke estimates for two further species, Black Grouse *Lyrurus tetrix* and Barn Owl *Tyto alba,* are described in table footnotes.

6. Wintering populations of waterbirds

Estimates of wintering waterbirds in GB were taken from Frost *et al.* (2019b). These were based on a variety of data sources, including the Wetland Bird Survey (WeBS), Goose & Swan Monitoring Programme (GSMP), Non-estuarine Coastal Waterbird Survey (NEWS), Winter Gull Roost Survey (WinGS), county bird reports, and for Mandarin Duck a novel analysis based on *Bird Atlas 2007–11* data. For full details see Frost *et al.* (2019b). Note that estimation methods have changed for many species and thus not all estimates are directly comparable with *APEP 3*. Most of these estimates relate to the period 2012/13 to 2016/17 (for estimates derived from WeBS or NEWS) or to 2011/12 to 2014/15 (for estimates derived from county reports).

UK estimates were produced by summing GB and NI estimates, the latter mostly from Burke *et al.* (2018) and generally relating to the period 2011/12 to 2015/16. However, estimates for several (mostly naturalised and scarcer) species were not published in Burke *et al.*, and for these species, NI estimates have been taken from other sources. Where no suitable source was found, the NI estimate is based on BirdTrack/BirdGuides/BTO survey data (see method 7). Details of the derivation are shown in the sources.

7. Wintering population estimates for scarce winter visitors based on BirdTrack/BirdGuides

Winter estimates are presented for a small

number of scarce non-breeding species that now winter annually in small numbers. The Panel developed a standardised approach to generate the *APEP 3* estimates for these species, to enable changes in status to be tracked in the future. The same method was used to produce *APEP 4* estimates. Records are extracted from the BirdTrack database, which also includes all records from the bird information service BirdGuides, and the highest count in each 10-km square was used to estimate GB and UK totals per month/year to calculate peak monthly totals each winter. For more details see Musgrove *et al.* (2013).

8. Estimates from *Winter Atlas 1981–84*

APEP 3 presented winter estimates for five species dating back to *Winter Atlas 1981–84* (Lack 1986); the same estimates for these species are included in this report given the absence of more recent assessments. These are: Fieldfare *Turdus pilaris*, Redwing *T. iliacus,* Black Redstart *Phoenicurus ochruros*, Brambling *Fringilla montifringilla* and Snow Bunting *Plectrophenax nivalis*.

9. Miscellaneous passage estimates

An estimate of passage numbers is presented only for Aquatic Warbler, which is included in view of its globally threatened status and its Birds Directive Annex I status requiring the classification of Special Protection Areas (of which there are four). The estimate is taken from Holt *et al.* (2018).

Assessing reliability of estimates

As with the two previous APEP reports, the Panel gave a broad reliability score to each estimate, ranging from 1 (for good) to 3 (for poor) for the date (or date range) indicated. In producing these scores, both the means of derivation of the original estimates and any methods used to extrapolate those original estimates were considered. As a general rule, estimates with a reliability score of 1 are based on direct counts with a minimum of extrapolation; those with a reliability score of 2 have been arrived at through extrapolation from reliable figures, or have a small amount of uncertainty around the estimate; and those with a reliability score of 3 were based on assumptions and opinion in place of actual fieldwork. Importantly, estimates scored as having poor reliability are still considered the best estimates available at present and are, in most cases, priorities for future work.

The reliability scores depend heavily on the means of derivation. For example, estimates of many common breeding passerines are based on trend-based extrapolation of estimates presented in *Breeding Atlas 1988–91*, which were themselves largely based on density measures derived from 1982 CBC data. Such estimates (indicated as '1–CBC/BBS' in the tables) are based on a foundation of sound data but have involved a large degree of extrapolation; consequently, most estimates derived by this method are given a reliability score of 2.

It is important to note that the reliability scores state how reliable the estimate was considered to be *for the date/period that it relates to*. In a number of cases, the estimates presented are now known to be either too high or too low *for the current population*, but suitable data sources were not available for satisfactory population updates. For example, the breeding estimate for Puffin *Fratercula arctica* has a reliability score of 1, since it is a good estimate of the population size for the period 1998–2002, when it was produced, but it is now believed to overestimate the size of the current population (Harris & Wanless 2011). In cases where an estimate is known to be too low (but where there is no basis of a more precise, larger estimate) it is marked '+', whereas an estimate known to be too high is marked '-'.

Results and Discussion

Population estimates for species of GB and UK birds are presented in table 1. Population estimates for selected subspecies or biogeographical populations are presented in table 2. Within the breeding season, the units are mostly 'pairs' or 'territories', depending on the survey methodology used (and for most purposes the two terms can be used interchangeably), while outside the breeding season all estimates relate to individuals. For some species, population size is more meaningfully estimated in other units, as set out in table 1. Estimates are rounded to two-and-a-half significant figures (i.e. rounded to three significant figures, with the third figure rounded to either five or a zero); thus a raw

estimate of 52,651 is rounded to 52,500 and one of 52,751 is rounded to 53,000. Estimates in previous APEP papers were rounded to two significant figures. The difference between UK and GB populations is the population size in Northern Ireland; readers should note that rounding may exaggerate this in some instances and reduce it in others.

In order to document the derivation of each estimate, the section number of the Methods (1, 2, etc.) is given against each estimate. For those estimates that involved forward extrapolation, the trend-measure used is also given; for example, '1–BBS' means extrapolation from a *Breeding Atlas 1988–91* estimate using the BBS trend (see table 1 key for other trend measures).

Footnotes are provided for estimates that require further explanation beyond that provided above, in particular where estimates were judged of lower reliability. References are given to document original sources. To avoid confusion, users of population estimates are strongly encouraged to cite the original published source of an estimate, other than for those calculated specifically for this paper.

The population estimates presented here indicate that there are now thought to be about 84 million breeding pairs of birds in the UK, similar to the number estimated by *APEP 3*. The Wren *Troglodytes troglodytes* continues to be the most common species and has increased slightly since *APEP 3*, with the current estimate of 11 million pairs making up about one in eight of our breeding birds. Out of a total of 249 breeding species, 20 species have estimates exceeding one million pairs (compared with 23 species in *APEP 3*) and these 20 contribute 77% of the total, with 58% provided by the ten commonest species alone. Although the total number of breeding pairs has fallen, the changes for individual species were mixed and showed slightly more increases than decreases.

The vast majority of these increases and decreases relate to genuine population changes measured by long-term monitoring schemes such as BBS for many of the commoner species, or by single-species surveys or RBBP data for scarcer species. However, some changes to estimates are the result of new species surveys or a change in the estimation method, particularly for wintering estimates (see Frost *et al.* 2019b), while some increases or decreases since *APEP 3* may represent better understanding of a species' abundance rather than genuine population change. For example, our breeding estimate for the non-native Canada Goose population is based on the winter population estimate, which has decreased in GB from 190,000 to 160,000 individuals as a result of methodological



Chris Knights/BTO

34. Golden Eagles *Aquila chrysaetos*, Scotland, May 2013. The Golden Eagle is one species for which a new population estimate has been produced using a single-species survey (Hayhow *et al.* 2017), and for the first time the UK population has gone above 500 pairs.

change (Musgrove *et al.* 2011; Frost *et al.* 2019b), yet the ten-year winter population trend for Canada Goose (2006/07 to 2016/17) is +10% (Frost *et al.* 2019a).

There have been relatively few such changes since *APEP 3* for the breeding season, with around 10% of breeding estimates being based on a change of method. In one case (Snow Bunting), the estimate of 60 pairs given here, following a national survey (Hayhow *et al.* 2018a), is coincidentally exactly the same as in *APEP 3*, but our confidence in the accuracy of the new estimate is markedly higher, since the previous figure was based on a tentative estimate published in *Birds of Scotland* (Forrester *et al.* 2007). Of the 10% of species for which a method change has occurred, around half update the previous estimates using trend data from BBS or SMP, and in most other cases the direction of change since *APEP 3* matches independent knowledge about the species trend. Canada Goose (see above) is the only species for which we can be fairly certain that the direction of change since *APEP 3* does not represent a genuine population change, although there is uncertainty in a handful of other cases.

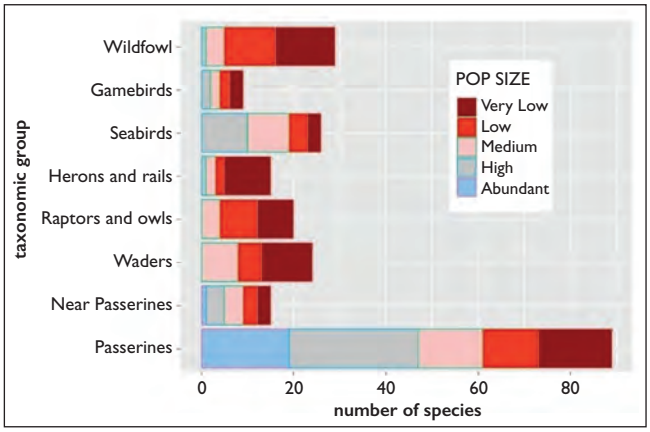


Fig. 1. Number of species in different taxonomic groups for which the breeding population estimates presented in this paper are: Very Low (<1,000 pairs), Low (1,000–9,999), Medium (10,000–99,999), High (100,000–999,999) and Abundant (1,000,000+).

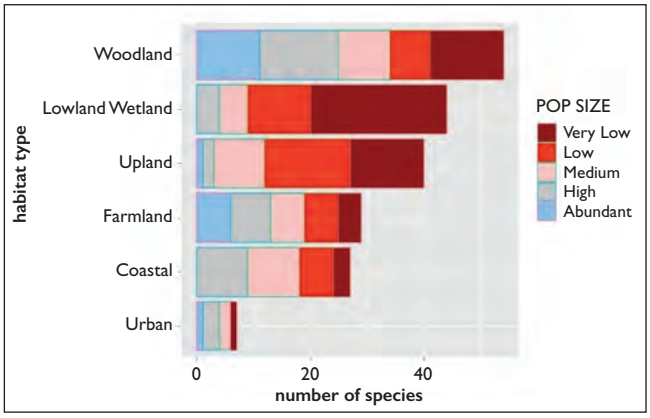


Fig. 2. Number of species using different preferred habitats for which the breeding population estimates presented in this paper are: Very Low (<1,000 pairs), Low (1,000–9,999), Medium (10,000–99,999), High (100,000–999,999) and Abundant (1,000,000+). Habitat types from Gibbons *et al.* 1993, with recent colonists (e.g. Little Egret *Egretta garzetta*) added to the most appropriate category.

There have been a larger number of method changes for the new GB wintering waterbird population estimates, but a comparison of species for which methodological changes did not occur suggests that numbers of many (but not all) geese have increased and the numbers of many (but not all) waders have decreased since *APEP 3* (Frost *et al.* 2019b). Of the other scarcer wintering species included in APEP for which new estimates have been produced, the population estimates have increased for ten species and decreased for six. Passage estimates are produced only for the Aquatic Warbler: estimates for this species have decreased in each of the four APEP reports, from a high of 40 individuals in *APEP 1* (relating to 1987–92) to three here.

Analysing the *APEP 4* breeding population estimates by taxonomic group (fig. 1) shows that the vast majority of species with populations greater than 100,000 breeding pairs are passerines, and all but one of the 20 species with more than a million pairs are passerines (the exception being Wood Pigeon *Columba palumbus*). This partly reflects the fact that most species found in the UK are passerines,

and that this group includes many ecological 'generalists', able to make use of widespread habitats across the UK, including woodland and gardens. Also, most passerines are small-bodied and exhibit particular ecological traits such as smaller territories and higher densities (Peters 1983).

The UK's coastline is important for breeding seabirds, with populations of international importance for some species (Mitchell *et al.* 2004). The seabirds include a notable proportion of species with a breeding population estimate of greater than 100,000 pairs, with the population estimate for Common Guillemot *Uria aalge* falling just under a million pairs (950,000).

Among the other groups, it is unsurprising that the breeding population estimates for raptors and owls are all below 100,000 pairs, since these species are apex predators and occur naturally in much lower numbers than their prey species. The UK is important for many wintering waders and wildfowl, the majority of which breed farther north, meaning that UK breeding populations for this group are less significant. However, for some species the UK breeding populations are important and are also in decline: although the current breeding population estimates for all wader species are below 100,000, those for Lapwing and Oystercatcher *Haematopus ostralegus* were both above this threshold in *APEP 3*, as was that for Curlew in *APEP 2*.

The breeding population sizes can also be compared across species habitat preferences (fig. 2). For this, we used the habitat-preference categories in Gibbons *et al.* (1993), excluding species not classified there but adding recent colonists to the most appropriate category if they show an obvious habitat preference. Note that each species is assigned to the preferred habitat rather than to all habitats in which it may occur; hence mildly woodland-preferring species such as Dunnock *Prunella modularis* may occur in overall larger numbers in farmland, which covers more than 70% of the UK's land surface.

Fig. 2 confirms that a mix of species with low and high population estimates can be found across all habitat types as predicted by ecological theory (e.g. Gaston & Blackburn 2000). Most species with breeding estimates of more than a million pairs are associated with woodland or lowland farmland. This reflects the ubiquity of these habitats, particularly farmland, in the UK. Moreover, some of the species categorised as woodland may be ecological generalists or occupy wooded features in other habitats such as hedgerows in farmland and gardens in urban areas.

Urban habitats are an exception to the pattern. Of the nine species found predominantly in urban habitats, only the Black Redstart has a UK population estimate of less than 10,000 breeding pairs. Most other species using urban habitats are generalists, abundant in other habitats.

Both upland and lowland wetland habitats support a larger proportion of species with fewer than 10,000 breeding pairs than other habitats. In the case of lowland wetlands, this may reflect the relative scarcity of certain wetland habitat such as reedbeds, and also that many recent colonists are found in lowland wetlands. In the uplands, the explanation for species with small population sizes includes generally low bird densities, combined with the relative scarcity of some upland habitats such as high mountains.

Recommendations

Progress since *APEP 3*

APEP 3 included 12 recommendations for improving avian population estimates, with the stated aim of encouraging efforts to improve reliability and have 'fewer 3s and more 1s in the next APEP report'. This aim has not been achieved. There are slightly fewer '1s' in this assessment, in part because some seabird population estimates are now based on extrapolation rather than census data. The reliability score for these species should improve in the near future with the completion of the 2015–20 seabird census. Although the levels of reliability have not changed substantially for non-seabirds, some progress has been made on some of the recommendations made in *APEP 3*. These should lead to further improvements in the quality of estimates in future, although this will need continued investment:

- The option to record the method of detection (song/call/visual) was introduced in BBS in 2014 and is now recorded by 79%

of BBS surveyors (Harris *et al.* 2019). This may enable distance-based population estimates (Method 2) to be refined in future, to account for differences in detectability of males and females for some species and thus allow improved estimates for additional species.

- BBS has also encouraged the surveying of additional upland squares by introducing the option to carry out one-off BBS surveys in remote squares ('Upland Rovers'). This should improve knowledge of population trends for upland species, enable trends for additional species to be reported and, in time and if sufficient squares are covered, may also enable population estimates to be calculated or improved for some species using distance sampling.
- Several papers about local population studies have been published, including on Firecrest *Regulus ignicapilla* (Clements *et al.* 2017) and Hobby *Falco subbuteo* (Clements *et al.* 2016), which have proposed national population estimates based on the local population densities. Although these estimates are not used in *APEP 4*, further work on these and other species is encouraged. Hewson *et al.* (2018) discussed some of the difficulties in using surveys of localised species to produce national population estimates. The Panel is considering providing guidance for local, single-species surveys, accounting for representativeness and how they can most usefully contribute towards national estimates. Additional local studies will also help to refine local density estimates that may help calibrate Atlas data.
- Work has been undertaken to improve the wintering population estimates for widely dispersed waterbirds such as Eurasian Teal *Anas crecca*, Mallard *A. platyrhynchos* and Moorhen *Gallinula chloropus*, for which a large proportion of the population is not counted by WeBS (Méndez *et al.* 2015). Frost *et al.* (2019b) took this into account when producing estimates for seven species.
- National winter bird surveys following a BBS-type methodology were carried out for the first time in 2018/19. The aim of this project was to assess usage of farmland habitats in England but there are currently no plans for ongoing monitoring. If resources become available, future monitoring of this sort on a wide scale could help achieve one of the *APEP 3* recommendations and improve our knowledge of the population of widely dispersed non-waterbird species during winter, which is currently poor.



Mark Rayment

35. The Black Redstart *Phoenicurus ochruros* remains the UK's only scarce breeding species that is found predominantly in urban areas (see fig. 2); Bedfordshire, February 2018.

Additional recommendations

Although progress has been made for some of the recommendations made in *APEP 3*, more work is required. The following list, which is not exhaustive, includes some recommendations repeated from *APEP 3* on which little or no progress has been made but which we still consider important.

- In recent years, repeat surveys for a number of scarcer species have not been conducted owing to lack of funding, including European Nightjar *Caprimulgus europaeus*, Dartford Warbler *Sylvia undata*, and Red-throated and Black-throated Divers *Gavia arctica*. Without repeat surveys we are reliant on ageing estimates for APEP and, more importantly, we do not have robust information to inform conservation prioritisation and (if required) conservation action for these species.
- Data on the breeding populations of a number of ducks remains poor and bespoke surveys are required for species including Common Shelduck *Tadorna tadorna*, Mandarin Duck, Gadwall, Teal, Eider, Red-breasted Merganser *Mergus serrator* and Goosander *M. merganser*.
- Targeted, species-specific surveys would also provide more robust estimates for other species for which current knowledge remains poor, including non-breeding Jack Snipe *Lymnocryptes minimus*, Hen Harrier and Merlin *Falco columbarius*; breeding and non-breeding Water Rail *Rallus aquaticus* and Common Snipe; and breeding Ptarmigan *Lagopus muta*, Common Swift *Apus apus*, Common Kingfisher *Alcedo atthis*, Sand Martin *Riparia riparia* and Rock Pipit.
- Information on the numbers of seabirds and other marine waterbirds using British waters outside the breeding season are still poorly known. As highlighted by Frost *et al.* (2019b), commercial data do exist for some locations but were not available to incorporate into population estimates; a clear mechanism is required to ensure that such data can be used.
- The UK holds internationally important wintering numbers of several gull species, and robust annual monitoring of wintering gulls at key sites, augmented by more comprehensive roost-site counts every few years, would improve understanding of population changes.
- Estimating the breeding populations of some seabirds is complicated by the increased use of other habitats such as urban areas. Work under way on urban gulls may help but this is also important for Great Cormorant *Phalacrocorax carbo* and some terns.

At the time of writing, the UK is scheduled to leave the EU, which means that the existing Article 12 reporting process to the EU will no longer apply. It is anticipated that directive obligations will be translated into domestic policy where there is an obligation to produce country environmental reports via JNCC; however, it will be important to ensure that any future changes to the reporting protocols are considered carefully and continue to be coordinated at a UK level. This should ensure that the UK can continue to meet other international obligations that require population estimates, and can continue to feed into BirdLife International's six-year assessment of the state of Europe's birds (which necessarily incorporates data from non-EU as well as EU countries). It will also be important that we continue to understand the status of bird populations in the UK in order to inform national conservation policy, and prioritise and undertake any necessary conservation actions. We thus anticipate that APEP updates on population size assessments will continue on a six-year cycle.

Acknowledgments

The majority of the estimates in this paper are based on work carried out by thousands of volunteers over many years, as surveyors, local scheme organisers or county bird recorders; their contribution cannot be underestimated. In addition, we acknowledge the support for the principal monitoring schemes given by the BTO, Department of Agriculture, Environment and Rural Affairs, Northern Ireland, Joint Nature Conservation Committee, Natural England, Natural Resources Wales, RSPB, Scottish Natural Heritage, Scottish Ornithologists' Club, Welsh Ornithological Society and the Wildfowl & Wetlands Trust; we also acknowledge the support and help from our colleagues at BirdWatch Ireland, and the provision of records from BirdGuides. Finally, particular help and advice was provided by Dawn Balmer, Andy Brown, Neil Calbrade, Rob Clements, Greg Conway, Allan Drewitt, Tim Dunn, Simon Foster, Simon Gillings, Hugh Hanmer, Sarah Harris, Richard Hearn, Mark Holling, Kerry Leonard, Patrick Lindley, Dario Massimino, Andy Musgrove,

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Additional sources of information:
[82] RBBP reports 2013–17 (Holling *et al.* 2015, 2016, 2017b, 2018, 2019);
[83] IRBBP reports 2012–16 (Perry 2013; Perry & Newton 2014; Newton 2015, 2016);
[84] BirdTrack database;
[85] www.projectparakeet.co.uk;
[86] Hugh Thurgate *in litt.*;
[87] bespoke calculation by N. J. Aebischer.

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Table 1. Population estimates of full species in categories A–C of the British List.

Key: **Season:** B = Breeding; P = Passage; W = Wintering. **Unit:** AOS = Apparently Occupied Sites; F = females; I = individuals; M = males; N = nests; P = pairs; T = territories. **Estimates:** a single figure or a range is given, but in some cases a mean with 95% confidence intervals in parentheses. **+/-:** population known to be larger (+) or smaller (-) than the estimate listed, but no better estimate available. **Der (Derivation):** Numbers correspond to the appropriate section under Derivation in the text; additional text indicates that the original source estimate has been extrapolated as described in the text, using one or more of the following trends (see text for further details): BBS = Breeding Bird Survey; CBC = Common Birds Census; NGC/BBS = National Gamebag Census; SMP = Seabird Monitoring Programme; WBS/WBBS = Waterways Bird Survey/Waterways Breeding Bird Survey. * indicates that a change since *APEP 3* may relate partly or wholly to the use of a different estimation method or improved knowledge rather than being a genuine change. **Rel (Reliability):** 1 (good) to 3 (poor); see text for further details. **Note:** Numbers refer to the footnotes of the table. **Reference:** Numbers correspond to those given in the References section.

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Capercaillie *Tetrao urogallus*	W	I	1,100	1,100		2015/16	5	1		76
Black Grouse *Lyrurus tetrix*	B	M	4,850	4,850		2016	5	1	1	87
Ptarmigan *Lagopus muta*	B	P	2,000–15,000	2,000–15,000		2007	5	2		23
Red Grouse *Lagopus lagopus*	B	P	265,000	265,000		2016	1–NGC/BBS	3		27, 1
Red-legged Partridge *Alectoris rufa*	B	T	72,500	72,500		2016	1–CBC/BBS	2		27
Grey Partridge *Perdix perdix*	B	T	37,000	37,000		2016	1–CBC/BBS	2		27, 83
Common Quail *Coturnix coturnix*	B	M	350	355		2013–17	3	2		82
Common Pheasant *Phasianus colchicus*	B	F	2,300,000	2,350,000		2016	1–CBC/BBS	2		70
Golden Pheasant *Chrysolophus pictus*	B	M	15	15		2010–14	3	3	2	44
Lady Amherst's Pheasant *Chrysolophus amherstiae*	B	M	0	0		2016	3	1	3	46
Brent Goose *Branta bernicla*	W	I	105,000	135,000		2012/13–2016/17	6	1		26, 9
Canada Goose *Branta canadensis*	B	P	54,000	54,500		2013–17	5*	3	4, 5	26, 84
	W	I	160,000	165,000		2012/13–2016/17	6*	2		26, 84
Barnacle Goose *Branta leucopsis*	B	P	1,450	1,550		2012–15	5	3	4, 6	26, 86
	W	I	105,000	105,000		–	6	1	7	26, 9
Snow Goose *Anser caerulescens*	B	P	2	2	+	2010–14	3	3		44
	W	I	75	75		2011/12–2015/16	6	2		26
Greylag Goose *Anser anser*	B	P	47,000	47,000		2013–17	5	3	4, 8	26, 7
	W	I	230,000	230,000		2012/13–2016/17	6	2		26, 9
Taiga Bean Goose *Anser fabalis*	W	I	230	230		2016/17–2017/18	6	1		26
Pink-footed Goose *Anser brachyrhynchus*	W	I	510,000	510,000	-	2015–16	6	1		59, 60, 26
Tundra Bean Goose *Anser serrirostris*	W	I	300	300		2011/12–2014/15	6	1		26

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
White-fronted Goose *Anser albifrons*	B	P	0–1	0–1		2013–17	3	1	9	47
	W	I	13,500	14,000		–	6	1	7	26, 9
Mute Swan *Cygnus olor*	B	P	6,500 (5,850–7,100)	7,000 (6,300–7,600)		2016	5–BBS	2		75
	W	I	50,500	52,500		2012/13–2016/17	6*	2		26, 9
Bewick's Swan *Cygnus columbianus*	W	I	4,350	4,350		2015	6	1		26, 9
Whooper Swan *Cygnus cygnus*	B	P	24	28		2013–17	3	1		82
	W	I	16,000	19,500		2015	6	1		30, 26, 9
Egyptian Goose *Alopochen aegyptiaca*	B	P	1,850	1,850		2013–17	5*	3	4	26
	W	I	5,600	5,600		2012/13–2016/17	6*	2		26
Common Shelduck *Tadorna tadorna*	B	P	7,600	7,850	+	2016	1–CBC/BBS*	3	10	27
	W	I	47,000	51,000		2012/13–2016/17	6	1		26, 9
Mandarin Duck *Aix galericulata*	B	P	4,400	4,400	+	2007–11	5*	3	4	26, 83
	W	I	13,500	13,500		2007–11	6*	3		26
Garganey *Spatula querquedula*	B	P	105	105		2013–17	3	2		47
Shoveler *Spatula clypeata*	B	P	1,100	1,100		2013–17	3	2		47
	W	I	19,000	19,500		2012/13–2016/17	6	1		26, 9
Gadwall *Mareca strepera*	B	P	1,250–3,150	1,250–3,200	+	2016	3–BBS	2	11	41, 83
	W	I	31,000	31,000		2012/13–2016/17	6	1		26, 9
Eurasian Wigeon *Mareca penelope*	B	P	200	200	+	2013–17	3*	3	12	47
	W	I	445,000	450,000		2012/13–2016/17	6*	1		26, 9
American Wigeon *Mareca americana*	W	I	17	18		2012/13–2016/17	7	3		84
Mallard *Anas platyrhynchos*	B	P	59,000–140,000	61,000–145,000	+	2016	5–CBC/BBS	3		3
	W	I	665,000	675,000		2012/13–2016/17	6	3		26, 9
Pintail *Anas acuta*	B	P	27	27		2013–17	3	1		47
	W	I	19,500	20,000		2012/13–2016/17	6	1		26, 9
Eurasian Teal *Anas crecca*	B	P	2,700–4,750	2,700–4,750		2016	1–CBC/BBS	3		27
	W	I	430,000	435,000		2012/13–2016/17	6*	2		26, 9
Green-winged Teal *Anas carolinensis*	W	I	32	34		2012/13–2016/17	7	3		84
Red-crested Pochard *Netta rufina*	B	P	39 (20–47)	39 (20–47)	+	2010–14	3	3	13	44
	W	I	570	570		2012/13–2016/17	6	1		26
Common Pochard *Aythya ferina*	B	P	695	720		2013–17	3	2		82
	W	I	23,000	29,000		2012/13–2016/17	6	1		26, 9
Ferruginous Duck *Aythya nyroca*	W	I	8	9		2012/13–2016/17	7	3		84

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Ring-necked Duck *Aythya collaris*	W	I	16	18		2012/13–2016/17	7	3		84
Tufted Duck *Aythya fuligula*	B	P	16,000–18,000	16,500–19,000		2016	1–CBC/BBS	3		69
	W	I	130,000	140,000		2012/13–2016/17	6*	1		26, 9
Greater Scaup *Aythya marila*	B	P	(0–1)	(0–1)		2013–17	3	2	14	82
	W	I	3,900	6,400	+	2011/12–2014/15	6	2		26, 9
Common Eider *Somateria mollissima*	B	P	36,000	37,000		2012–15	5*	2	15	26, 56
	W	I	81,500	86,000		2011/12–2014/15	6*	2		26, 9
Surf Scoter *Melanitta perspicillata*	W	I	23	24		2012/13–2016/17	7	3		84
Velvet Scoter *Melanitta fusca*	W	I	3,350	3,350	+	2011/12–2014/15	6*	2		26, 84
Common Scoter *Melanitta nigra*	B	P	52	52		2007	5	1		21
	W	I	135,000	135,000	+	2011/12–2014/15	6*	3		26, 9
Long-tailed Duck *Clangula hyemalis*	W	I	13,500	13,500	+	2011/12–2014/15	6*	2		26, 84
Common Goldeneye *Bucephala clangula*	B	F	200	200		2006–10	3	2	16	41
	W	I	18,500	21,000		2012/13–2016/17	6*	2		26, 9
Smew *Mergellus albellus*	W	I	125	125		2012/13–2016/17	6	1		26
Goosander *Mergus merganser*	B	P	4,800 (4,250–5,250)	4,800 (4,250–5,250)	+	2016	5–WBS/WBBS	2		29, 83
	W	I	14,500	14,500		2012/13–2016/17	6*	2		26, 84
Red-breasted Merganser *Mergus serrator*	B	P	1,550 (1,350–1,750)	1,650	+	2008–11	5*	3	17	49, 4
	W	I	10,500	11,000		2012/13–2016/17	6*	2		26, 9
Ruddy Duck *Oxyura jamaicensis*	B	P	2–3	2–3		2017	5	1		36
	W	I	23	23		2016	5	1		36
European Nightjar *Caprimulgus europaeus*	B	M	4,600 (3,700–5,500)	4,600 (3,700–5,500)		2004	5	1		14
Common Swift *Apus apus*	B	P	59,000 (43,000–75,000)	59,000 (43,000–75,000)	+	2016	2–BBS	3		65
Great Bustard *Otis tarda*	B	P	4	4		2013–17	3	1	18	47
Common Cuckoo *Cuculus canorus*	B	P	17,000 (8,950–24,500)	18,000 (9,800–26,000)	+	2016	2–BBS	3		65
Rock Dove *Columba livia*	B	P	460,000 (375,000–545,000)	465,000 (380,000–550,000)		2016	2–BBS	2		65
Stock Dove *Columba oenas*	B	T	320,000	320,000		2016	1–CBC/BBS	2		27
Wood Pigeon *Columba palumbus*	B	P	5,050,000 (4.75–5.35m)	5,150,000 (4.85–5.45m)		2016	2–BBS	2		65
Turtle Dove *Streptopelia turtur*	B	T	3,600	3,600		2016	1–CBC/BBS	2		27
Collared Dove *Streptopelia decaocto*	B	P	795,000 (715,000–875,000)	810,000 (730,000–890,000)		2016	2–BBS	2		65

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Water Rail *Rallus aquaticus*	B	T	3,900	3,900	+	2016–17	5	2	19	24
Corn Crake *Crex crex*	B	M	1,100	1,100		2013–17	3	1		47
Baillon's Crake *Porzana pusilla*	B	M	0–6	0–6		2012	5	3	20	43
Spotted Crake *Porzana porzana*	B	M	27	27		2013–17	5	1	21	47
Moorhen *Gallinula chloropus*	B	T	200,000	210,000		2016	1–CBC/BBS	2	22	27
	W	I	300,000	305,000		2012/13–2016/17	6	3		26, 63
Common Coot *Fulica atra*	B	P	25,500	26,000	+	2016	1–CBC/BBS	3		27
	W	I	200,000	205,000		2012/13–2016/17	6*	1		26, 9
Common Crane *Grus grus*	B	P	31	31		2013–17	3	1		47
	W	I	175	175		2017/18	6	2		26
Little Grebe *Tachybaptus ruficollis*	B	P	3,300–6,650	3,650–7,300		2016	1–WBS/WBBS	2		27
	W	I	15,000	15,500		2012/13–2016/17	6	2		26, 9
Red-necked Grebe *Podiceps grisegena*	B	P	(0–1)	(0–1)		2013–17	3	1		82
	W	I	59	60		2011/12–2014/15	6	2		26, 84
Great Crested Grebe *Podiceps cristatus*	B	P	4,300	4,900	+	2016	1–BBS	2		27
	W	I	16,500	18,000		2012/13–2016/17	6*	1		26, 9
Slavonian Grebe *Podiceps auritus*	B	P	28	28		2013–17	3	1		47
	W	I	920	995		2011/12–2014/15	6	2		26, 84
Black-necked Grebe *Podiceps nigricollis*	B	P	55	55		2013–17	3	1		47
	W	I	115	115		2012/13–2016/17	6	1		26
Stone-curlew *Burhinus oedicnemus*	B	P	365	365	+	2013–17	3	1		47
Oystercatcher *Haematopus ostralegus*	B	P	92,500	95,500		2016	5–BBS	2		67
	W	I	285,000	305,000		2012/13–2016/17	6	1		26, 9
Black-winged Stilt *Himantopus himantopus*	B	P	3 (0–6)	3 (0–6)		2013–17	3	1		82, 2
Avocet *Recurvirostra avosetta*	B	P	1,950	1,950		2013–17	3	1		47
	W	I	8,700	8,700		2012/13–2016/17	6	1		26
Northern Lapwing *Vanellus vanellus*	B	P	96,500	97,500		2016	5–BBS	2	23	67, 13
	W	I	620,000	635,000		2006/07	6	2		28, 9
European Golden Plover *Pluvialis apricaria*	B	P	32,500–50,500	32,500–50,500		2016	5–BBS	2	24	73
	W	I	400,000	410,000		2006/07	6	2		28, 9
Grey Plover *Pluvialis squatarola*	W	I	33,500	33,500		2012/13–2016/17	6	1		26, 9
Ringed Plover *Charadrius hiaticula*	B	P	5,300 (5,100–5,500)	5,450 (5,250–5,600)		2007	5	1		16
	W	I	41,500	42,500		2012/13–2016/17	6*	1		26, 9

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Little Ringed Plover *Charadrius dubius*	B	P	1,250 (1,200–1,300)	1,250 (1,200–1,300)		2007	5	1		16
Dotterel *Charadrius morinellus*	B	M	425 (280–645)	425 (280–645)		2011	5	2		32
Whimbrel *Numenius phaeopus*	B	P	310	310		2009	5	1		51
	W	I	38	41		2012/13–2016/17	6	1		26, 84
Eurasian Curlew *Numenius arquata*	B	P	58,000	58,500		2016	5–BBS	2	23	67, 13
	W	I	120,000	125,000		2012/13–2016/17	6	1		26, 9
Bar-tailed Godwit *Limosa lapponica*	W	I	50,500	53,500		2012/13–2016/17	6*	1		26, 9
Black-tailed Godwit *Limosa limosa*	B	P	53	53		2013–17	3	1		47
	W	I	39,000	41,000		2012/13–2016/17	6*	1		26, 9
Turnstone *Arenaria interpres*	W	I	40,000	43,000		2012/13–2016/17	6	1		26, 9
Red Knot *Calidris canutus*	W	I	265,000	265,000		2012/13–2016/17	6	1		26, 9
Ruff *Calidris pugnax*	B	F	13	13		2013–17	3	1		47
	W	I	895	920		2012/13–2016/17	6	1		26, 84
Temminck's Stint *Calidris temminckii*	B	P	0	0		2013–17	3	1		82
Sanderling *Calidris alba*	W	I	20,000	20,500		2012/13–2016/17	6	1		26, 9
Dunlin *Calidris alpina*	B	P	8,600–10,500	8,600–10,500		2005–07	5	2		23, 8, 54
	W	I	345,000	350,000		2012/13–2016/17	6	1		26, 9
Purple Sandpiper *Calidris maritima*	B	P	1	1		2013–17	3	1		47
	W	I	9,700	9,900		2012/13–2016/17	6	2		26, 9
Little Stint *Calidris minuta*	W	I	8	8		2012/13–2016/17	6	1		26
Woodcock *Scolopax rusticola*	B	M	55,000 (42,000–69,000)	57,000 (43,000–71,000)		2016	5	1	25	38
	W	I	1,400,000	1,400,000		2003/04	6	3		62, 63, 26
Jack Snipe *Lymnocryptes minimus*	W	I	100,000	110,000		2004/5	6	3		62, 63, 26
Common Snipe *Gallinago gallinago*	B	P	64,500	66,500		2016	5–BBS	3	23	67, 13
	W	I	1,000,000	1,100,000		2004/5	6	3		62, 63, 26
Red-necked Phalarope *Phalaropus lobatus*	B	M	64	64		2013–17	3	1		47
Common Sandpiper *Actitis hypoleucos*	B	P	13,000	13,000		2016	5–WBS/WBBS	2		19
	W	I	52	52		2012/13–2016/17	6*	1		26
Green Sandpiper *Tringa ochropus*	B	P	2	2		2013–17	3	1		47
	W	I	290	290		2011/12–2014/15	6*	3		26, 84
Common Redshank *Tringa totanus*	B	P	22,000	22,000		2016	5–BBS	2	23	67, 13
	W	I	94,500	100,000		2012/13–2016/17	6	1		26, 9

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Wood Sandpiper *Tringa glareola*	B	P	30	30		2013–17	5	1		47
Spotted Redshank *Tringa erythropus*	W	I	67	68		2012/13–2016/17	6	1		26, 84
Greenshank *Tringa nebularia*	B	P	1,100	1,100	+	1995	5	2		31
	W	I	810	920		2012/13–2016/17	6	1		26, 9
Kittiwake *Rissa tridactyla*	B	P	195,000 (170,000–250,000)	205,000 (175,000–255,000)		2015	4–SMP	2		61
Black-headed Gull *Chroicocephalus ridibundus*	B	P	130,000	140,000	+	1998–2002	4	1		61
	W	I	2,200,000 (2.1–2.2m)	2,200,000 (2.1–2.3m)	+	2003/04–2005/06	6	2		62, 63, 26
Mediterranean Gull *Ichthyaetus melanocephalus*	B	P	1,200	1,200	+	2013–17	3	2		47
	W	I	4,000	4,000		2011/12–2014/15	6	2		26, 84
Common Gull *Larus canus*	B	P	48,000	48,500		1998–2002	4	1		61
	W	I	700,000 (670,000–720,000)	710,000 (680,000–730,000)	+	2003/04–2005/06	6	2		62, 63, 26
Ring-billed Gull *Larus delawarensis*	W	I	17	21		2012/13–2016/17	7	3		84
Great Black-backed Gull *Larus marinus*	B	P	15,000 (7,200–19,000)	15,000 (7,200–19,000)	-	2015	4–SMP	2		61
	W	I	76,000 (71,000–81,000)	77,000 (72,000–82,000)	+	2003/04–2005/06	6	2		62, 63, 26
Glaucous Gull *Larus hyperboreus*	W	I	155	165	+	2011/12–2015/16	6	2		26, 84
Iceland Gull *Larus glaucoides*	W	I	330	355	+	2011/12–2015/16	6	2		26, 84
Herring Gull *Larus argentatus*	B	P	130,000	130,000	-	1998–2002	4	1	26	61
	W	I	730,000 (700,000–760,000)	740,000 (710,000–780,000)	+	2003/04–2005/06	6	2		62, 63, 26
Caspian Gull *Larus cachinnans*	W	I	125	125		2011/12–2014/15	6	2		26, 84
Yellow-legged Gull *Larus michahellis*	B	P	2	2		2013–17	3	1		47
	W	I	840	840		2011/12–2014/15	6	2		26, 84
Lesser Black-backed Gull *Larus fuscus*	B	P	110,000	110,000	-	1998–2002	4	1		61
	W	I	120,000 (120,000–130,000)	130,000 (120,000–130,000)	+	2003/04–2005/06	6	2		62, 63, 26
Sandwich Tern *Thalasseus sandvicensis*	B	P	12,500 (11,500–14,000)	14,000 (13,000–15,000)		2015	4–SMP	2		61
	W	I	53	65		2011/12–2014/15	6	2		26, 84
Little Tern *Sternula albifrons*	B	P	1,450	1,450		2013–17	4	1		82
Roseate Tern *Sterna dougallii*	B	P	100	100		2013–17	3	1		47
Common Tern *Sterna hirundo*	B	P	9,600 (7,550–11,500)	11,000 (8,900–13,500)		2015	4–SMP	2		61
Arctic Tern *Sterna paradisaea*	B	P	52,500	53,500		2000	4	1		61
Great Skua *Stercorarius skua*	B	P	9,650	9,650	+	1998–2002	4	1	27	61
Arctic Skua *Stercorarius parasiticus*	B	P	785 (535–1,550)	785 (535–1,550)		2015	4–SMP	2	28	61

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Common Guillemot *Uria aalge*	B	P	885,000	950,000		1998–2002	4	1		61
Razorbill *Alca torda*	B	P	140,000 (93,000–215,000)	165,000 (100,000–250,000)		2015	4–SMP	2		61
Black Guillemot *Cepphus grylle*	B	P	19,000	19,500		1998–2003	4	1		61
Puffin *Fratercula arctica*	B	P	580,000	580,000	-	1998–2002	4	1		61
Red-throated Diver *Gavia stellata*	B	P	1,250 (1,000–1,550)	1,250 (1,000–1,550)		2006	5	1		17
	W	I	21,500	21,500	+	2011/12–2014/15	6*	3	29	26, 9
Black-throated Diver *Gavia arctica*	B	P	215 (190–250)	215 (190–250)		2006	5	1		20
	W	I	560	560	+	2004/05–2008/09	6	3		62, 26
Great Northern Diver *Gavia immer*	W	I	4,350	4,400	+	2015/16	6*	3		26, 9
White-billed Diver *Gavia adamsii*	W	I	80	80	+	2010–12	6	2		26
European Storm-petrel *Hydrobates pelagicus*	B	AOS	25,500 (21,000–33,500)	25,500 (21,000–33,500)		1998–2002	4	2		61
Leach's Storm-petrel *Oceanodroma leucorhoa*	B	P	48,000 (36,500–65,000)	48,000 (36,500–65,000)		1998–2002	4	2		61
Fulmar *Fulmarus glacialis*	B	P	350,000 (195,000–680,000)	350,000 (195,000–680,000)		2015	4–SMP	2		61
Manx Shearwater *Puffinus puffinus*	B	P	295,000 (280,000–315,000)	300,000 (280,000–320,000)		1998–2002	4	2		61
Northern Gannet *Morus bassanus*	B	N	295,000	295,000		2013–14	4	1		53
Shag *Phalacrocorax aristotelis*	B	P	17,500 (13,500–20,500)	17,500 (13,500–20,500)		2015	4–SMP	2		61
	W	I	110,000	110,000	+	1998–2002	6	2		62, 26
Great Cormorant *Phalacrocorax carbo*	B	P	8,200	8,900		1998–2002	4	1	30	61
	W	I	62,000	64,500	+	2012/13–2016/17	6*	2		26, 9
Glossy Ibis *Plegadis falcinellus*	W	I	27	27		2011/12–2014/15	6	1		26
Eurasian Spoonbill *Platalea leucorodia*	B	P	29	29		2017	3	1	31	47
	W	I	105	105		2011/12–2014/15	6	1		26
Eurasian Bittern *Botaurus stellaris*	B	M	191	191	+	2017	5	1		47
	W	I	795	795		2017/18	5	2		26
Little Bittern *Ixobrychus minutus*	B	M	5	5		2013–17	3	1		47
Night Heron *Nycticorax nycticorax*	B	P	0–1	0–1		2013–17	3	1	32	47
Cattle Egret *Bubulcus ibis*	B	P	10–15	10–15		2017	3	1	31	47
	W	I	65	66		2011/12–2014/15	6	2		26, 84
Grey Heron *Ardea cinerea*	B	P	9,950	10,500 (10,000–11,000)		2013–17	5	1	33	79
	W	I	45,000	45,500		2012/13–2016/17	6*	2		26, 9
Purple Heron *Ardea purpurea*	B	P	0	0		2013–17	3	1	34	82

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Great White Egret *Ardea alba*	B	P	8–12	8–12		2017	3	1	31	47
	W	I	72	72		2011/12–2014/15	6	2		26, 84
Little Egret *Egretta garzetta*	B	P	1,100	1,100	+	2013–17	3	2	35	82
	W	I	11,500	11,500		2012/13–2016/17	6*	2		26, 9
Osprey *Pandion haliaetus*	B	P	240	240	+	2013–17	3*	2	36	47
Honey-buzzard *Pernis apivorus*	B	P	33–69	33–69		2000	5	2	37	5, 68
Golden Eagle *Aquila chrysaetos*	B	P	510	510		2015	5	1		33
Eurasian Sparrowhawk *Accipiter nisus*	B	P	28,500	30,500		2016	1–CBC/BBS	2		66
Northern Goshawk *Accipiter gentilis*	B	P	620	620	+	2013–17	3	2		47
Marsh Harrier *Circus aeruginosus*	B	P	590–695	590–695		2016	3–BBS	2	38	40
Hen Harrier *Circus cyaneus*	B	P	500	545		2016	5	1		81
Montagu's Harrier *Circus pygargus*	B	P	8	8		2013–17	3	1		47
Red Kite *Milvus milvus*	B	P	4,350	4,400		2016	3–BBS	2	11	41, 83
White-tailed Eagle *Haliaeetus albicilla*	B	P	122	123	+	2017	3	1		47
Rough-legged Buzzard *Buteo lagopus*	W	I	29	29		2012/13–2016/17	7	3		84
Common Buzzard *Buteo buteo*	B	P	61,500–85,000	63,000–87,500		2016	5–BBS	2		11
Barn Owl *Tyto alba*	B	P	4,000–14,000	4,000–14,000		2016	5*	3	39	74
Tawny Owl *Strix aluco*	B	P	50,000	50,000		2005	5	2		25
Little Owl *Athene noctua*	B	P	3,600 (2,350–4,900)	3,600 (2,350–4,900)		2016	2–BBS	2		65
Long-eared Owl *Asio otus*	B	P	1,600–5,300	1,800–6,000		2007–11	5	3		71
Short-eared Owl *Asio flammeus*	B	P	610–2,150	620–2,200		2007–11	5	3		27
Common Kingfisher *Alcedo atthis*	B	P	3,650–6,100	3,850–6,400		2016	1–WBS/WBBS	3		27
European Bee-eater *Merops apiaster*	B	P	1 (0–3)	1 (0–3)		2013–17	3	1	40	82
Wryneck *Jynx torquilla*	B	P	0	0		2013–17	3	2	41	47
Lesser Spotted Woodpecker *Dryobates minor*	B	P	600–1,000	600–1,000		2015	3	3	42	45
Great Spotted Woodpecker *Dendrocopos major*	B	P	130,000 (120,000–145,000)	130,000 (120,000–145,000)		2016	2–BBS	2		65, 83
Green Woodpecker *Picus viridis*	B	P	45,500 (40,500–50,500)	45,500 (40,500–50,500)		2016	2–BBS	2		65
Common Kestrel *Falco tinnunculus*	B	P	30,000	31,000		2016	1–CBC/BBS	2		27
Merlin *Falco columbarius*	B	P	1,150 (850–1,450)	1,150 (890–1,450)		2008	5	2		22

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Hobby *Falco subbuteo*	B	P	2,050	2,050		2016	5–BBS	2	43	10
Peregrine Falcon *Falco peregrinus*	B	P	1,650 (1,500–1,800)	1,750 (1,600–1,900)		2014	5	1	44	78
Rose-ringed Parakeet *Psittacula krameri*	B	P	12,000	12,000		2016	5–BBS	2	45	85
Red-backed Shrike *Lanius collurio*	B	P	3	3		2013–17	3	1		47
Great Grey Shrike *Lanius excubitor*	W	I	98	98		2012/13–2016/17	7	3		84
Golden Oriole *Oriolus oriolus*	B	M	0 (0–2)	0 (0–2)		2013–17	3	1	46	82
Eurasian Jay *Garrulus glandarius*	B	T	165,000	170,000		2016	1–CBC/BBS	2		27
Magpie *Pica pica*	B	T	550,000	610,000		2016	1–CBC/BBS	2		27
Red-billed Chough *Pyrrhocorax pyrrhocorax*	B	P	330	335		2014–15	5	1		35
	W	I	1,250	1,250		2014/15	5	1		35
Jackdaw *Coloeus monedula*	B	P	1,450,000 (1.3–1.65m)	1,550,000 (1.35–1.75m)		2016	2–BBS	2		65
Rook *Corvus frugilegus*	B	P	885,000 (775,000–1,000,000)	980,000 (865,000–1,100,000)		2016	5–BBS	2		57
Carrion Crow *Corvus corone*	B	T	1,050,000	1,050,000		2016	1–CBC/BBS	2		27
Hooded Crow *Corvus cornix*	B	T	185,000	285,000		2016	1–BBS	2		27
Common Raven *Corvus corax*	B	P	9,500	10,000	+	2016	1–BBS	3		27
Waxwing *Bombycilla garrulus*	W	I	9,750	10,000		2012/13–2016/17	7	3		84
Coal Tit *Periparus ater*	B	T	590,000	660,000		2016	1–CBC/BBS	2		27
Crested Tit *Lophophanes cristatus*	B	P	1,000–2,000	1,000–2,000		2007	5	2		23
Marsh Tit *Poecile palustris*	B	T	28,500	28,500		2016	1–CBC/BBS	2		27
Willow Tit *Poecile montanus*	B	P	2,750	2,750	-	2016	1–CBC/BBS	2	47	27
Blue Tit *Cyanistes caeruleus*	B	T	3,250,000	3,400,000		2016	1–CBC/BBS	2		27
Great Tit *Parus major*	B	T	2,200,000	2,350,000		2016	1–CBC/BBS	2		27
Bearded Tit *Panurus biarmicus*	B	P	695	695		2013–17	3	2		47
Woodlark *Lullula arborea*	B	P	2,300 (1,850–2,750)	2,300 (1,850–2,750)		2016	5–BBS	2	48	15
Skylark *Alauda arvensis*	B	T	1,500,000	1,550,000		2016	1–CBC/BBS	2		27
Shore Lark *Eremophila alpestris*	W	I	110	110		2012/13–2016/17	7	3		84
Sand Martin *Riparia riparia*	B	N	64,500–210,000	70,500–225,000		2016	1–WBS/WBBS	3		27
Barn Swallow *Hirundo rustica*	B	T	625,000	705,000		2016	1–CBC/BBS	2		27

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
House Martin *Delichon urbicum*	B	P	470,000 (330,000–610,000)	480,000 (335,000–620,000)		2016	2–BBS	2		65
Cetti's Warbler *Cettia cetti*	B	M	3,450	3,450	+	2016	3	2		46
Long-tailed Tit *Aegithalos caudatus*	B	T	370,000	380,000		2016	1–CBC/BBS	2		27
Wood Warbler *Phylloscopus sibilatrix*	B	M	6,500 (6,000–7,050)	6,500 (6,000–7,050)		2016	1–BBS	3		6
Yellow-browed Warbler *Phylloscopus inornatus*	W	I	25	25		2012/13–2016/17	7	3		84
Willow Warbler *Phylloscopus trochilus*	B	T	2,050,000	2,300,000		2016	1–BBS	2		27
Common Chiffchaff *Phylloscopus collybita*	B	T	1,650,000	1,750,000		2016	1–CBC/BBS	2		27
Iberian Chiffchaff *Phylloscopus ibericus*	B	P	0–1	0–1		2013–17	3	1	49	82, 50
Aquatic Warbler *Acrocephalus paludicola*	P	I	3	3	+	2013–17	9	3		48
Sedge Warbler *Acrocephalus schoenobaenus*	B	T	220,000	240,000		2016	1–CBC/BBS	2		27
Reed Warbler *Acrocephalus scirpaceus*	B	P	130,000 (100,000–155,000)	130,000 (100,000–155,000)	+	2016	2–BBS	2		65, 83
Marsh Warbler *Acrocephalus palustris*	B	P	8	8		2013–17	3	1		47
Icterine Warbler *Hippolais icterina*	B	P	0–2	0–2		2013–17	3	1	50	82
Grasshopper Warbler *Locustella naevia*	B	T	9,750	12,000		2016	1–BBS	3		27
Savi's Warbler *Locustella luscinioides*	B	P	5	5		2013–17	3	1		47
Blackcap *Sylvia atricapilla*	B	T	1,600,000	1,650,000		2016	1–CBC/BBS	2		27
Garden Warbler *Sylvia borin*	B	T	145,000	145,000		2016	1–CBC/BBS	2		27
Lesser Whitethroat *Sylvia curruca*	B	T	79,000	79,000		2016	1–CBC/BBS	2		27
Common Whitethroat *Sylvia communis*	B	T	1,100,000	1,100,000		2016	1–CBC/BBS	2		27
Dartford Warbler *Sylvia undata*	B	P	2,200	2,200		2017	3	2	51	47
Firecrest *Regulus ignicapilla*	B	T	2,000	2,000	+	2016	3	3	52	46
Goldcrest *Regulus regulus*	B	T	675,000	790,000		2016	1–BBS	2		27
Wren *Troglodytes troglodytes*	B	T	9,750,000	11,000,000		2016	1–CBC/BBS	2		27
Eurasian Nuthatch *Sitta europaea*	B	T	250,000	250,000		2016	1–CBC/BBS	2		27
Eurasian Treecreeper *Certhia familiaris*	B	T	210,000	225,000		2016	1–CBC/BBS	2		27
Common Starling *Sturnus vulgaris*	B	P	1,650,000 (1.45–1.8m)	1,750,000 (1.55–1.95m)		2016	2–BBS	2		65
Ring Ouzel *Turdus torquatus*	B	P	7,300 (5,550–9,400)	7,300 (5,550–9,400)		2016	5–BBS*	2	53	80
Blackbird *Turdus merula*	B	P	4,850,000 (4.6–5.05m)	5,050,000 (4.8–5.25m)		2016	2–BBS	2		65

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Fieldfare *Turdus pilaris*	B	P	0–1	0–1		2013–17	3	2	54	82
	W	I	680,000	720,000		1981–84	8	3		55
Redwing *Turdus iliacus*	B	P	24	24		2013–17	3	3		47
	W	I	650,000	690,000		1981–84	8	3		55
Song Thrush *Turdus philomelos*	B	T	1,200,000	1,300,000		2016	1–CBC/BBS	2		27
Mistle Thrush *Turdus viscivorus*	B	T	150,000	165,000		2016	1–CBC/BBS	2		27
Spotted Flycatcher *Muscicapa striata*	B	T	38,500	41,500		2016	1–CBC/BBS	3		27
Robin *Erithacus rubecula*	B	T	6,650,000	7,350,000		2016	1–CBC/BBS	2		27
Bluethroat *Luscinia svecica*	B	P	0–1	0–1		2013–17	3	1	55	82
Common Nightingale *Luscinia megarhynchos*	B	M	5,550 (5,100–6,000)	5,550 (5,100–6,000)		2012	5*	2	56	39
Pied Flycatcher *Ficedula hypoleuca*	B	P	22,000–25,000	22,000–25,000		2016	1–BBS	3		27
Black Redstart *Phoenicurus ochruros*	B	P	58	58		2013–17	3	2		47
	W	I	400	400		1981–84	8	3		55
Common Redstart *Phoenicurus phoenicurus*	B	P	135,000 (97,000–170,000)	135,000 (97,000–170,000)		2016	2–BBS	2		65
Whinchat *Saxicola rubetra*	B	P	49,500 (19,500–79,000)	49,500 (19,500–79,000)		2016	2–BBS	2		65, 83
European Stonechat *Saxicola rubicola*	B	P	61,000 (39,500–83,000)	65,000 (43,000–87,000)		2016	2–BBS	2		65
Northern Wheatear *Oenanthe oenanthe*	B	P	165,000 (115,000–220,000)	170,000 (120,000–220,000)		2016	2–BBS	2		65
Dipper *Cinclus cinclus*	B	P	6,350–19,000	6,900–20,500		2016	1–WBS/WBBS	2		27
House Sparrow *Passer domesticus*	B	P	5,150,000 (4.65–5.65m)	5,300,000 (4.8–5.75m)		2016	2–BBS	2		65
Tree Sparrow *Passer montanus*	B	T	225,000	245,000		2016	1–BBS	2		27
Dunnock *Prunella modularis*	B	T	2,350,000	2,500,000		2016	1–CBC/BBS	2		27
Yellow Wagtail *Motacilla flava*	B	T	19,500	19,500		2016	1–CBC/BBS	2		27
Grey Wagtail *Motacilla cinerea*	B	P	33,500	37,000		2016	1–WBS/WBBS	3		27
Pied Wagtail *Motacilla alba*	B	P	495,000 (435,000–560,000)	505,000 (445,000–570,000)		2016	2–BBS	2		65
Meadow Pipit *Anthus pratensis*	B	P	2,250,000 (1.95–2.55m)	2,450,000 (2.1–2.75m)		2016	2–BBS	2		65
Tree Pipit *Anthus trivialis*	B	P	105,000 (66,000–145,000)	105,000 (66,000–145,000)		2016	2–BBS	2		65
Water Pipit *Anthus spinoletta*	W	I	205	205		2012/13–2016/17	7	3		84
Rock Pipit *Anthus petrosus*	B	P	34,000	36,000		1988–1991	1	3		27
Common Chaffinch *Fringilla coelebs*	B	T	4,800,000	5,050,000		2016	1–CBC/BBS	2		27

Species	Season	Unit	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Brambling *Fringilla montifringilla*	B	P	0–1	0–1		2013–17	3	3		82
	W	I	45,000–1,800,000	45,000–1,800,000		1981–84	8	3		55
Hawfinch *Coccothraustes coccothraustes*	B	P	500–1,000	500–1,000		2011	5*	3		12
Bullfinch *Pyrrhula pyrrhula*	B	T	225,000	265,000		2016	1–CBC/BBS	2		27
Greenfinch *Chloris chloris*	B	P	760,000 (710,000–810,000)	785,000 (735,000–835,000)		2016	2–BBS	2		65
Twite *Linaria flavirostris*	B	P	7,800 (5,800–10,000)	7,850 (5,850–10,000)		2013	5	2		77
Linnet *Linaria cannabina*	B	T	530,000	560,000		2016	1–CBC/BBS	2		27
Common Redpoll *Acanthis flammea*	B	P	12	12		2013–17	3	2		47
	W	I	335	335		2012/13–2016/17	7	3		84
Lesser Redpoll *Acanthis cabaret*	B	P	220,000	260,000		2016	1–BBS	3		27
Arctic Redpoll *Acanthis hornemanni*	W	I	9	11		2012/13–2016/17	7	3		84
Parrot Crossbill *Loxia pytyopsittacus*	B	P	65	65		2008	5	2		72
Scottish Crossbill *Loxia scotica*	B	P	6,800 (4,050–11,500)	6,800 (4,050–11,500)		2008	5	2		72
Common Crossbill *Loxia curvirostra*	B	P	25,000 (19,000–33,000)	26,000 (19,500–34,000)		2016	2–BBS	2		65
Goldfinch *Carduelis carduelis*	B	P	1,600,000 (1.4–1.75m)	1,650,000 (1.45–1.8m)		2016	2–BBS	2		65
European Serin *Serinus serinus*	B	P	0	0		2013–17	3	1		82
Siskin *Spinus spinus*	B	P	430,000	445,000		2016	1–BBS	3		27
Lapland Bunting *Calcarius lapponicus*	B	P	0–1	0–1		2013–17	3	1		82
	W	I	310	310		2012/13–2016/17	7	3		84
Snow Bunting *Plectrophenax nivalis*	B	T	60 (48–83)	60 (48–83)		2011	5	2		34
	W	I	9,000–13,500	10,000–15,000		1981–84	8	3		55
Corn Bunting *Emberiza calandra*	B	T	11,000 (9,050–13,000)	11,000 (9,050–13,000)		2016	5–CBC/BBS	2		18
Yellowhammer *Emberiza citrinella*	B	T	685,000	700,000		2016	1–CBC/BBS	2		27
Cirl Bunting *Emberiza cirlus*	B	T	1,100	1,100		2016	5	1	57	52
Reed Bunting *Emberiza schoeniclus*	B	T	255,000	275,000		2016	1–CBC/BBS	2		27

Table 1. Population estimates of full species in categories A–C of the British List – Notes

1. Estimate comes from a bespoke calculation by N. J. Aebischer using the most recent survey information available from ongoing annual monitoring in Scotland and Wales, adjusted using the 2005 survey data (Sim *et al.* 2008) to account for areas not covered by ongoing monitoring, and data from the 2014 survey in England (Warren *et al.* 2015), extrapolated to 2016 using the average annual change since the previous survey.
2. A large reduction in numbers has occurred and this non-native species is believed to be on the way to extinction in the UK.
3. A mean of four males were present during 2010–14 (Holling *et al.* 2014a, 2017a), but the last male has not been seen since May 2016 (Holling *et al.* 2018) and the species is now believed to be extinct as a breeding species in the UK.
4. GB estimates based on the wintering estimates for the naturalised population divided by 3.
5. The NI estimate for Canada Goose is based on BirdTrack records since there is no published estimate for NI, applying the same method used for scarce winter visitors (method 7).
6. The NI estimate uses a count from Strangford Lough, which is believed to be the only feral population of Barnacle Geese in NI (H. Thurgate, S. Wolsey pers. comm.).
7. Dates for the different subspecies estimates are shown in table 2.
8. Note that the NI estimate (264 individuals, 88 pairs) is based on the latest published figure (Boland & Crowe 2008) and is known to be an underestimate. There are now flocks of at least 300 in Belfast in July and possibly 700–800 or more in the whole of NI (K. Mackie pers. comm.) but there is no definitive recent count or estimate to replace the published figure.
9. A feral pair bred in Essex or northeast London in 2017 (Holling *et al.* 2019).
10. The BBS extrapolation uses the UK anchor trend, which is thought to better reflect the UK trend than the England joint trend used in *APEP 3*. The large decrease in the population estimate since *APEP 3* results from this change of estimation method rather than a substantial decline in breeding numbers: the species is believed to have been increasing in England since *APEP 3*.
11. This species is no longer considered by the RBBP, so the previous GB estimate from RBBP data has been updated by extrapolation using BBS.
12. The RBBP five-year mean is believed to be an underestimate and the population may fall closer to or within the range of 300–500 given by previous APEP papers. However, the previous estimate dates from 1968–72 and is of uncertain reliability so is replaced here with the more recent RBBP estimate to enable comparison with future RBBP monitoring.
13. The difference between the wintering and breeding estimates suggests that either (i) breeding is substantially under-recorded or (ii) there is either a large non-breeding population or a substantial influx from Europe during winter. RBBP recorded 67 breeding pairs in 2017 (Holling *et al.* 2019).
14. A single brood was recorded in 2014. It could have involved a hybrid pair, since males were not seen with the brood, although males were seen at the same site earlier in 2014 (Holling *et al.* 2016).
15. The GB estimate repeats the method used in *APEP 3*, and takes the wintering estimate minus birds wintering between Lincolnshire and Essex (assumed to be Dutch breeders), adjusted by a scale of 2.21 to calculate the number of breeding pairs. Note that the NI estimate (Leonard 2010) is an estimate of breeding females rather than breeding pairs.
16. The *APEP 3* estimate is repeated rather than using more recent RBBP data; the latter are believed to be incomplete following more limited monitoring since 2011.
17. The GB estimate is based on a similar method to that used for *APEP 3*, based on *1988–91 Atlas* data, but uses *2008–11 Atlas* data. Note that Humphreys *et al.* (2016) gave some caveats and state that the estimates should be considered a minimum. The UK estimate makes the assumption that breeding density per occupied 10-km square in NI is the same as in GB.
18. RBBP figures show that there were seven nesting females in 2016 in the reintroduced population, but the five-year mean figure has been used rather than the most recent year, since it is unclear whether this level will be sustained.
19. Francis *et al.* (2020) give a range of 2,500–3,900 pairs as the estimate, but state that 3,900 is the best single estimate for the UK population and believe that the true figure may be higher.
20. A total of 6–11 singing males were reported during the national Spotted Crake survey in 2012 (Ausden *et al.* 2013), although RBBP subsequently reported 0–6 since some records were not submitted to BBRC and hence remained unverified. It remains unclear whether 2012 was a one-off or whether more regular undetected breeding attempts may occur.
21. The estimate of 28 singing males from the survey in 2012 (Schmitt *et al.* 2015) gives confidence that the RBBP data are likely to be complete or near-complete.
22. Burke *et al.* (2018) did not give an estimate for wintering Moorhens, so the UK figure uses the *APEP 3* estimate for NI, based on data from 2004/05–2008/09.
23. The NI estimate comes from a survey in 2013 (Colhoun *et al.* 2015), which is added to the extrapolated GB estimate in order to produce the UK figure.
24. The previous estimate has been updated by extrapolation using the BBS trend since 1995. Note that the previous estimate covered the period 1980–2000 so the extrapolation may not account for changes that occurred in the 1980s.
25. The 2013 survey covered GB only, so the UK estimate presented here makes the assumption that the population change in NI matched the rate of decline observed in GB.
26. Note that although the *APEP 3*/*Seabird 2000* estimate is repeated here, an error in the UK total in *APEP 3* has been corrected (birds from the IoM were included in the UK total in *APEP 3*).
27. The sample from the SMP was not considered sufficiently robust to use the trend to update the population estimate and therefore the *Seabird 2000* estimate is again repeated. Perkins *et al.* (2018) stated that there has been little overall change in the number of territories; they suggested a slight (but not statistically significant) increase in the area they covered (Orkney, Shetland and Handa).
28. Perkins *et al.* (2018) give an estimate of 554 (422–728) breeding pairs for Orkney, Shetland and Handa in 2015, with the area they covered holding around 90% of the population during *Seabird 2000*. The estimate presented here, based on SMP trends, would fall just inside the upper confidence interval given by Perkins *et al.* assuming similar declines across all colonies.

29. This estimate is repeated from Frost *et al.* (2019b), who suggested that it may be a substantial underestimate.

30. The previous estimate, from *Seabird 2000*, is repeated here. SMP trends since then suggest a decline, but these have not been used to produce an updated estimate owing to uncertainty about whether they are fully representative of both coastal and inland colonies, for which the trends are different.

31. Following rapid recent increases, the most recent available annual figure has been used rather than a five-year mean, since breeding numbers seem likely to be sustained or to increase further in the future.

32. A pair bred in 2017 in Somerset, the first record of breeding in the UK (Holling *et al.* 2019).

33. The UK estimate comes from the BTO's Heronries Census published in BirdTrends (Woodward *et al.* 2018) with the calculation accounting for both actual counts and sites not surveyed. Recent census coverage in NI has been near-complete so the actual counts from NI were deducted from the UK estimate to derive the GB estimate.

34. This species first nested in the UK in 2010 but nesting was not confirmed during 2013–17.

35. Now that this species is relatively common and widespread, breeding is likely to be under-recorded and this may be a substantial underestimate. A comparison with the wintering estimate suggests a breeding population of around 2,700 pairs or more, if the proportion breeding was similar to that for Grey Heron (but we do not have sufficient information to make this assumption).

36. The *APEP 3* estimate (200–250) was based on expert opinion rather than the RBBP five-year mean, which is likely to be an underestimate. Hence, although this estimate falls in the middle of the previous range, it is likely to be an underestimate and the population is believed to be still increasing. RBBP recorded 258 pairs in 2017 (Holling *et al.* 2019).

37. The previous estimate is repeated and is considered the best current estimate. The RBBP five-year mean of 39 for 2013–17 is at the lower end of this range and is believed to be an underestimate. Roberts & Law (2014) estimated 100–150 pairs; while their figures could be correct, it is felt that further survey work is required to confirm that this estimate is robust.

38. This uses the BBS trend to extrapolate from the last national survey, in 2005, which found 429 pairs (Holling *et al.* 2008).

39. The *APEP 3* figure of 4,000 pairs is believed to be an underestimate. Extrapolating using the BBS trend gives an updated estimate of 14,200 pairs but there are uncertainties about the reliability of using BBS for this crepuscular/nocturnal species. Expert opinion and information from ringing and nest record schemes suggests that the population falls somewhere between these two figures and hence a tentative estimate of 4,000–14,200 pairs is given here.

40. Two pairs nested in both 2014 and 2015 and three pairs in 2017 (RBBP).

41. Wryneck is judged to be extinct in the UK as breeding has not been confirmed since 2002 (Holling *et al.* 2019), although singing males are still recorded occasionally.

42. This species is difficult to find and breeding is believed to be under-recorded, so producing a robust population estimate is challenging. Casual recording suggests that the severe decline of this species has continued, so the revised estimate of 600–1,000 pairs (Holling *et al.* 2017b) is adopted.

43. The current estimate is based on an extrapolation from the *APEP 3* figure and reflects the decline recorded by BBS during this period. Clements & Everett (2012) and Clements *et al.* (2016) suggested that densities are much higher than previously thought in some areas and that 3,000–3,500 pairs should be a lower limit for the population, which is perhaps as high as 5,000 pairs. Further work will hopefully lead to a robust, revised population estimate.

44. Wilson *et al.* (2018) gave two estimates; this is the most conservative and the one considered to be their main estimate.

45. The *APEP 3* estimate from 2012 was updated using the BBS trend, reflecting a substantial increase over the four years to 2016.

46. This species is probably now extinct as a breeding bird in the UK (Holling *et al.* 2018).

47. RBBP data (196–678 pairs in 2016) suggest that the population may be lower than this estimate, but the species is probably under-recorded by RBBP. A survey being carried out in 2019 and 2020 may produce a better estimate.

48. The estimate from the 2006 survey was extrapolated using the BBS trend. Although the mean sample size (22 squares) is too low for a formal BBS trend, the data reinforces the expected pattern of a strong decline caused by the cold winters in 2008/09 and 2009/10 followed by a slight recovery. The extrapolated estimate is likely to be more reliable than repeating the 2006 survey estimate, carried out prior to the severe winter declines.

49. The estimate of 0–1 pairs assumes that the 2015 breeding record was most likely to be a one-off and assumes that singing males holding territory in most years (mean estimate of 2 (1–4) over the period 2013–17) are likely to be overshooting migrants. However, the recent records may be an indication that this species is in the process of colonising the UK.

50. This species remains an occasional breeder only. Two singing males were recorded in both 2015 and 2016 (RBBP).

51. The most recent national survey figure was 3,200 pairs in 2006 but numbers crashed during the severe winters in 2008/09 and 2009/10 (Clark & Eyre 2012). The 2017 RBBP total is considered to be a more reliable current figure, although note that it is based in part on county estimates, mostly within the core range in Hampshire.

52. Clements *et al.* (2017) suggested that the population is in excess of 4,000 singing males. However, this is based on extrapolation from core areas, not a full survey. Following the removal of Firecrest from the RBBP species list (Holling *et al.* 2019), and in the absence of evidence that breeding distribution is other than sparse away from core areas, the RBBP threshold of 2,000 is given as a minimum estimate. It is considered that the population is likely to be in excess of 2,000 pairs but that the data are not sufficiently robust to give a more precise estimate.

53. Note that the 2012 estimate includes a correction to account for detectability and the higher population estimate than *APEP 3* reflects this rather than a genuine population change. The BBS trend suggests little or no change since the 1999 survey, although the sample size during the early years of BBS was too low to report BBS trends.

54. The last confirmed breeding was in 2012 (Holling *et al.* 2017b).

55. One pair bred in 2016 (Ince 2018).

56. The estimate is the mean of 12 population estimates presented by Hewson *et al.* (2018), and the means of the confidence limits from these 12 estimates.

57. The estimate is the maximum estimate from the survey; this is considered to be the most accurate assessment of the population (Jeffs *et al.* 2018).

Table 2. Population estimates of subspecies or geographical populations. For key, see table 1.

Species	Season	Units	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Red Grouse *Lagopus lagopus hibernica*	B	P	0	200		2005	3	2		1
Red Grouse *Lagopus lagopus scotica*	B	P	265,000	265,000		2016	1–NGC/BBS	3		27
'Dark-bellied Brent Goose' *Branta b. bernicla*	W	I	98,500	98,500		2012/13–2016/17	6	1		26
'Pale-bellied Brent Goose' *Branta bernicla hrota* (Nearctic)	W	I	1,550	31,000		2012/13–2016/17	6	1		26, 9
'Pale-bellied Brent Goose' *Branta bernicla hrota* (Svalbard)	W	I	3,400	3,400		2012/13–2016/17	6	1		26
Barnacle Goose *Branta leucopsis* (Greenland)	W	I	56,000	56,000		Spring 2018	6	1		26
Barnacle Goose *Branta leucopsis* (Svalbard)	W	I	43,500	43,500		2017/18	6	1		26
Barnacle Goose *Branta leucopsis* (naturalised)	W	I	4,350	4,700		2011/12–2014/15	6	2		26, 86
'Greenland White-fronted Goose' *Anser albifrons flavirostris*	W	I	11,500	11,500		2017	6	1		26, 9
'European White-fronted Goose' *Anser a. albifrons*	W	I	2,100	2,100		2012/13–2016/17	6	1		26
Common Eider *Somateria mollissima* (excl. Shetland)	B	P	34,000	35,000		2012–15	5*	2	1	26, 56
Common Eider *Somateria mollissima* (excl. Shetland)	W	I	77,000	81,000		2011/12–2014/15	5*	2		26, 9
Common Eider *Somateria mollissima* (Shetland)	B	P	2,100	2,100		2015	6	1	2	37
Common Eider *Somateria mollissima* (Shetland)	W	I	4,600	4,600		2015	6	1		37
Rock Dove *Columba l. livia*	B	P	1,000–5,000	1,000–5,000		1988–91	5	2		23
Feral Pigeon *Columba livia domestica*	B	P	455,000 (375,000–540,000)	460,000 (375,000–545,000)		2016	2–BBS	2		65, 23
Black-tailed Godwit *Limosa limosa islandica*	B	P	7	7		2013–17	3	1		82
Black-tailed Godwit *Limosa l. limosa*	B	P	46	46		2013–17	3	1		82
Common Guillemot *Uria a. aalge*	B	P	850,000	850,000		1998–2002	4	1	3	61
Common Guillemot *Uria aalge albionis*	B	P	34,500	100,000		1998–2002	4	1	3	61
Great Cormorant *Phalacrocorax c. carbo*	B	P	6,450	7,150		2015	4	2		61, 64
Great Cormorant *Phalacrocorax c. carbo*	W	I	55,500	58,500		2012/13–2016/17	4	3		26, 9, 64
Great Cormorant *Phalacrocorax c. sinensis*	B	P	1,550	1,550		2012	5	2		64
Great Cormorant *Phalacrocorax c. sinensis*	W	I	6,100	6,100		2012/13	5	3		64

Species	Season	Units	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Eurasian Jay *Garrulus glandarius hibernicus*	B	T	0	4,250		2016	1–CBC/BBS	2		27
Coal Tit *Periparus ater britannicus*	B	T	590,000	590,000	+	2016	1–CBC/BBS	2	4	27
Coal Tit *Periparus ater hibernicus*	B	T	0	74,000	-	2016	1–CBC/BBS	2		27
Wren *Troglodytes troglodytes indigenus*	B	T	9,750,000	11,000,000		2016	1–CBC/BBS	2		27
'Fair Isle Wren' *Troglodytes troglodytes fridariensis*	B	T	39	39		2013–17	3	1		47
'Hebridean Wren' *Troglodytes troglodytes hebridensis*	B	T	5,000–10,000	5,000–10,000		2007	5	2		23
'St Kilda Wren' *Troglodytes troglodytes hirtensis*	B	T	230	230		2007	3	2		46
'Shetland Wren' *Troglodytes troglodytes zetlandicus*	B	T	1,500–3,000	1,500–3,000		2007	5	2		23
'Shetland Starling' *Sturnus vulgaris zetlandicus*	B	T	10,000–20,000	10,000–20,000		2007	5	2		23
'Hebridean Song Thrush' *Turdus philomelos hebridensis*	B	P	500–2,200	500–2,200		2003	5	2		58
Dipper *Cinclus cinclus gularis*	B	P	6,350–19,000	6,350–19,000	-	2016	1–WBS/WBBS	2	5	27
Dipper *Cinclus cinclus hibernicus*	B	P	?	570–1,700	+	2016	1–WBS/WBBS	2	6	27
Dunnock *Prunella modularis hebridium*	B	T	75,000–125,000	245,000–295,000		2016	5–CBC/BBS	3		23, 27
Yellow Wagtail *Motacilla flava flavissima*	B	T	19,500	19,500		2016	1–CBC/BBS	2		27
'Blue-headed Wagtail' *Motacilla f. flava*	B	P	0–1	0–1		2013–17	5–CBC/BBS	1		82
Pied Wagtail *Motacilla alba yarrellii*	B	P	495,000 (435,000–560,000)	505,000 (445,000–570,000)		2016	1–WBS/WBBS	2		27
White Wagtail *Motacilla a. alba*	B	P	2	2		2013–17	5	1	7	82

Notes

1. See table 1 for details of breeding calculation.
2. Derived from wintering population estimate, adjusted by a factor of 2.21.
3. Note that *U. aalge albionis* is calculated based on all Guillemots in Northern Ireland being *albionis;* Guillemots in SW Scotland north to, and including, Ailsa Craig are *albionis*; all Guillemots in England excluding Northumberland are *albionis*. All other Scottish birds and those in Northumberland are *U. a. aalge*.
4. Subspecies *brittanicus* mostly found in GB but also occurs in extreme northeast of Ireland where it intergrades into *hibernicus*; the precise nature of zone of intergradation is unknown.
5. This is an overestimate as some Scottish birds are assigned to *hibernicus*.
6. Estimate is for NI but is an underestimate as *hibernicus* also occurs in Hebrides and parts of west coast of mainland Scotland.
7. The RBBP five-year mean is stated as three breeding pairs (Holling *et al.* 2019) but includes mixed pairs, which are excluded here.

Table 3. Population estimates of non-native species for which breeding has been confirmed in the UK since 2011. For key, see table 1. Note that estimates for Category C species are in table 1.

Species	Season	Units	GB estimate	UK estimate	+/-	UK date	Der	Rel	Note	Reference
Helmeted Guineafowl *Numida meleagris*	B	P	1	1		2010–14	3	3		44
Indian Peafowl *Pavo cristatus*	B	P	4	4		2010–14	3	3	1	44
Wild Turkey *Meleagris gallopavo*	B	P	1	1		2010–14	3	3		42, 44
Bar-headed Goose *Anser indicus*	B	P	1	1		2010–14	3	3		44
Emperor Goose *Anser canagicus*	B	P	0–1	0–1		2010–14	3	3		44
Black Swan *Cygnus atratus*	B	P	19	19		2010–14	3	3		44
Ruddy Shelduck *Tadorna ferruginea*	B	P	1	1		2010–14	3	2		44
Wood Duck *Aix sponsa*	B	P	3	3		2010–14	3	3		44
Muscovy Duck *Cairina moschata*	B	P	10	10		2010–14	3	3		44
Harris's Hawk *Parabuteo unicinctus*	B	P	2	2		2010–14	3	3		44
Eagle Owl *Bubo bubo*	B	P	2	2		2010–14	3	1		44
Monk Parakeet *Myiopsitta monachus*	B	P	18	18		2010–14	3	2		44

Note

1. Cheke (2019) suggests some local UK populations of this species could be considered self-sustaining.

Table 4. Population estimates for six species with notable breeding populations in the Isle of Man. For key, see table 1. Estimates are shown here for the five species for which separate IoM estimates were published in *APEP 3*, and also for Little Tern.

Species	Season	Units	IoM estimate	+/-	UK date	Der	Rel	Note	Reference
Ringed Plover *Charadrius hiaticula*	B	P	147 (135–160)		2007	5	1		16
Herring Gull *Larus argentatus*	B	P	7,100	-	1998–2002	4	1		61
Little Tern *Sternula albifrons*	B	P	18		2013–17	3	1		82
Hen Harrier *Circus cyaneus*	B	P	30		2016	5	1		81
Peregrine Falcon *Falco peregrinus*	B	P	22 (22–23)		2014	5	1		78
Red-billed Chough *Pyrrhocorax pyrrhocorax*	B	P	160		2014–15	5	1		35

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In Practice

Bulletin of the Institute of Ecology and Environmental Management



Editorial

Greetings from the new IEEM President

Hello everyone, if you have been following your *In Practice*, you will know that I am now your President. So first, thank you for your support and secondly, we have an interesting and exciting journey to share over the next two years.

The year in the wings as President-elect has been very useful and interesting. What have I learnt in my apprenticeship? First, the organisation is you. It is financially dependant on you being members; you coming to conferences; you attending training courses. I am hoping to work with the Institute to provide added value for these – some of the services you most appreciate according to the recent membership survey.

The second lesson for me was seeing how much voluntary time people give the Institute to support something in which they believe passionately. These are all busy people, most have 'day' jobs as well, yet they sit on committees, write and re-write position statements, develop best practice guidance and discuss all the IEEM issues selflessly. IEEM depends on them for their creativity, time and commitment.

Then there is the Secretariat – small but committed and hard working. I am delighted to be working with your new CEO, Sally Hayns, in developing the next IEEM Strategy and Business Plan. More on that as it evolves early next year.

I suspect every new President comes to this job with great thoughts on what legacy they can leave after two years and how best to develop these grand ideas. So I will give you mine. I would like to:

- see an expansion and improvement in the Institute's services offered to members;
- see increasing recognition of the importance of IEEM membership and our professional status;
- diversify the income streams for improved financial stability;
- work towards developing our own chartered status (which also means an increase in membership);
- develop more best practice guidance/position statements on various ecological matters such as habitat creation, offsetting and living landscapes in line with the recent Lawton Report (have you read it yet? You should);
- increase the recognition of the environmental management part of our name; and
- develop a green strategy/guidance for IEEM.

There will be more every day things where I will be representing IEEM - press releases, workshops and meetings, letters to write and awards to present. I suspect that I have not appreciated the full scale of the job yet!

One of the most important things for my Presidency though is that IEEM is 21 in 2012. We are looking to develop a programme of events. If you have any ideas for your region or any national activities – let us know. Don't worry if they sound crazy – they should be fun, they should celebrate IEEM and our vision/mission, but they could be cricket matches or quizzes, field events or competitions, collaborative pond-digging events or gentle visits – you choose. Think about getting some sponsorship for IEEM as well.

This is me – but it is a two-way process. So what can you do to support me and IEEM? We want to provide more and better services to members but we do need to know what you want so you need to tell us. Come to conferences and attend training courses. Let us know if what we offer is not what you want and do get actively involved. What about running a workshop on your pet subject for us (assuming it is relevant!)?

The more volunteers we have to help run regional or national events/committees, the less responsibility there is on those already engaged. So offer your help to increase the robustness and representativeness of the Institute.

Come and talk to me and to the other Council and Committee members, e-mail us with your ideas, issues and solutions. We are interested in feedback. We can only take on board other issues if you tell us about them.

I look forward to meeting and talking to as many of you as possible during the next couple of years. I probably won't remember your names, but I will remember your faces!

Penny Anderson CEnv FIEEM
IEEM President

Wishing you all a happy Christmas and a prosperous New Year!
-- From everyone at IEEM --

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Cover image: Biodiversity Beyond 2010

Photography: Pete Johnstone CEnv MIEEM

Artwork on the cover will normally illustrate an article in, or the theme of, the current issue. The Editor would be pleased to consider any such material from authors.

Valuing Bats in Ecological Impact Assessment

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Since their publication in 2006, IEEM's *Guidelines on Ecological Impact Assessment in the UK* have provided a standard approach and structure for ecological impact assessments (EcIAs) which has been hugely beneficial. However, while the process of assessing impacts on habitats and species (receptors) is described in detail in the guidelines, the valuation of receptors, on which the rest of the assessment depends, was considered too complex to include. Valuation of receptors therefore remains entirely a matter of professional judgement for ecologists preparing EcIAs. This article provides a framework for valuing bat roosts, commuting routes and foraging areas which is, as far as possible, objective in the hope that increased consistency in this aspect of EcIA can also be achieved.

In our experience, bats can be amongst the hardest of receptors to value consistently. Their highly mobile nature, combined with a high level of legal protection and conservation concern (seven species are UK Biodiversity Action Plan priority species), means that roosting, commuting or foraging bats are likely to be a potential issue on almost any development site, but it does not follow that every site is necessarily valuable for bats.

It has been argued in some EcIAs that any valued ecological receptor (VER) relating to bats is of national or international importance due to their level of legal protection. This approach is erroneous, conflating legislative and policy matters with ecological principles.

This article is based on a paper (Wray *et al.* 2007) presented at the 'Advances in EcIA for Mammals' symposium run jointly by the Mammal Society, Zoological Society of London and IEEM.

Geographic Frames of Reference

In the EcIA guidelines, valuation involves assigning a receptor to a geographic frame of reference, *i.e.* International, UK/National, Regional, County, and District, Local or Parish so that the level of weight or importance attached to any impact can be appropriately assessed. Clearly, any robust approach to valuation needs to be underpinned by adequate survey data, as specified by the Bat Conservation Trust (2007) or other specific guidance, for example, for wind farms see Rodrigues *et al.* (2008) or Natural England (2009). In the case of bats as receptors, such data are not readily forthcoming and in this paper, we attempt to make a judgement as to the appropriate valuations.

Distribution and Rarity

One of the problems in assigning bat VERs to geographic frames of reference in EcIA is that the distribution and rarity of species varies greatly. A pipistrelle maternity colony and a grey long-eared maternity colony are not likely to be valuable at the same geographic scale, due the great difference in rarity of the two species. To overcome this, our first step is to define the relative rarity of each species as Common, Rarer, or Rarest based on the known distribution and population size in the UK. Table 1 shows this for each bat species for the UK regions. Accurate population figures are not available for all species even at this geographic scale and the data presented here are primarily estimates in Richardson (2000), Harris *et al.* (1995), and Harris and Yalden (2008).

Valuing Roosts

Bats use a range of different types of roost, and not all have the same level of importance in supporting a local population of bats. At one end of the scale, a night roost may be used by a single bat or small number of bats to rest during night-time feeding activities.

Table 1: Categorising bats by distribution and rarity

Rarity within range	England	Wales	Scotland	Northern Ireland
Rarest (popn. under 10,000)	greater horseshoe[1] Bechstein's[2] alcathoe?[3] greater mouse-eared[4] barbastelle[5] grey long-eared[6]	greater horseshoe whiskered[7] Brandt's[8] Bechstein's alcathoe? noctule[9] Nathusius' pipistrelle[10] serotine[11] barbastelle	whiskered Brandt's alcathoe? noctule Nathusisus' pipistrelle Leisler's[12]	whiskered
Rarer (popn. 10,000 – 100,000)	lesser horseshoe[13] whiskered Brandt's Daubenton's[14] Natterer's[15] Leisler's noctule Nathusius' pipistrelle serotine	lesser horseshoe Daubenton's Natterer's brown long-eared[16]	Daubenton's Natterer's brown long-eared	Daubenton's Natterer's Leisler's Nathusius' pipistrelle brown long-eared
Common (popn. over 100,000)	common pipistrelle[17] soprano pipistrelle[18] brown long-eared	common pipistrelle soprano pipistrelle	common pipistrelle soprano pipistrelle	common pipistrelle soprano pipistrelle

[1]*Rhinolophus ferrumequinum*; [2]*Myotis bechsteinii*; [3]*Myotis alcathoe*; [4]*Myotis myotis*; [5]*Barbastella barbastellus* ; [6]*Plecotus austriacus*; [7]*Myotis mystacinus*; [8]*Myotis brandtii*; [9]*Nyctalus noctula*; [10]*Pipistrellus nathusii*; [11]*Eptesicus serotinus*; [12]*Nyctalus leisleri*; [13]*Rhinolophus hipposideros*; [14]*Myotis daubentonii*; [15]*Myotis nattereri*; [16]*Plecotus auritus*; [17]*Pipistrellus pipistrellus*; [18]*Pipistrellus pygmaeus*

There may be limited fidelity to the roost structure and it could easily be replaced. At the other end of the scale, a large maternity roost or hibernation site where large numbers of bats gather from over a wide area and have used the same site over many years or generations. This type of roost is obviously of much greater significance.

Having categorised bat species by rarity and distribution above, different roost types can be assigned to a geographic frame of reference based on the rarity of the species concerned. Table 2 shows our proposed valuations for different roost types, for bats in each rarity category. So, for example, maternity sites of common species would be valued at County level, whereas maternity sites of the rarest species would be valued at National level. Conversely, feeding perches for even the rarest species would not be valued above County level, reflecting their lower importance for the integrity of bat populations.

Table 2: Valuing bat roosts

Geographic frame of reference	Roost types
District, Local or Parish	Feeding perches (common species) Individual bats (common species) Small numbers of non-breeding bats (common species) Mating sites (common species)
County	Maternity sites (common species) Small numbers of hibernating bats (common and rarer species) Feeding perches (rarer/rarest species) Individual bats (rarer/rarest species) Small numbers of non-breeding bats (rarer/rarest species)
Regional	Mating sites (rarer/rarest species) including well-used swarming sites Maternity sites (rarer species) Hibernation sites (rarest species) Significant hibernation sites for rarer/rarest species or all species assemblages
National/UK	Maternity sites (rarest species) Sites meeting SSSI guidelines
International	SAC sites

Valuing Commuting Routes and Foraging Areas

Two additional problems apply when assigning values to commuting and foraging bats. Firstly, rather than looking at different kinds of bat activity (*i.e.* different roost types), we are trying to distinguish between different intensities of use or bat behaviour. Secondly, these activities cannot be considered in isolation and need to be assessed in the context of surrounding roosting, commuting and foraging opportunities, which will often be outside the survey area for a project. For example, commuting bats close to a SSSI roost for that species should take that into account, even if the roost is not itself a receptor being considered in the EcIA. Similarly, valuation of bats foraging should take into account the availability of foraging habitat in the surrounding areas. In some situations, the type of foraging habitat available close to a roost can be critical to, for example, breeding success and recruitment and these factors must all be brought in to the analysis.

Table 3: Scoring system for valuing commuting and foraging bats

Geographic frame of reference	Score
International	>50
National	41 - 50
Regional	31 - 40
County	21 - 30
District, local or parish	11 - 20
Not important	1 - 10

For both commuting routes and foraging habitats, a scoring system has been developed to determine the appropriate geographic frame of reference (see Table 3). Usually several bat species are present on a site, in which case each species should be valued individually, and the highest value obtained (usually for the rarest species present) used in the EcIA.

Commuting Routes

When valuing commuting routes the rarity of the species involved, the approximate numbers of bats using them (based on survey data), the proximity of known roosts, and the nature and complexity of linear features in the landscape are all taken into account to put the bat activity recorded into context. Table 4 provides scores for each of these factors. One score is taken from each column, depending on the 'best fit' for the situation and they are added together in order to arrive at a total score.

For example, a soprano pipistrelle commuting route in England comprising small numbers of bats away from known roosts in a landscape of flailed hedgerows would score 2+10+4+3=19. Referring to Table 3 indicates that this commuting route would be of District value or below. However, large numbers of commuting barbastelle bats on the same site would score 20+20+4+3=47 indicating a commuting route of National value.

Foraging Areas

Similarly, when valuing foraging areas, the rarity of the species, the approximate number of bats using them (based on survey data), the proximity of known roosts, and the landscape-scale foraging opportunities available are all taken into account. Table 5 provides scores for each of these factors, using the same approach as described for commuting routes above.

For example, individual lesser horseshoe bats foraging in Wales, in an area with few potential roost sites and isolated woodland patches in an agricultural landscape, would score 5+5+3+3=16 indicating a foraging area of District value or below. Conversely, a small number of foraging serotine bats on the same site would score 20+10+3+3=36 indicating a foraging area of Regional value.

Using and Refining this Approach

One obvious refinement to the approach described would be further sub-division of the UK regions in Table 1, particularly for England and Scotland where the bat assemblages present in the south and north of the countries are very different. Currently, there are insufficient published data on regional bat population sizes to enable this, indeed even some of the national data are inferred or based on estimates of unknown accuracy.

Regular users of the IEEM guidelines will have noted that some of the geographic frames of reference have been combined in the tables. In our experience, National and UK value are rarely distinguishable when considering bat receptors. The does not mean that we fail to appreciate the regional distinctiveness of our bat populations, rather that a receptor of National value in any UK country will tend to be of UK value. The other geographic frames of reference combined are those at the lower end of the scale: District, Local and Parish. We have found it extremely difficult to provide a framework that can meaningfully distinguish between these, given that receptors falling into these categories are generally small numbers of common or individual rarer bats. However, the combining of these categories is less critical, given that the threshold for full consideration in the EcIA is usually set above this level.

Table 4: Valuing commuting routes

Species	Number of bats	Roosts/potential roosts nearby	Type and complexity of linear features
Common (2)	Individual bats (5)	None (1)	Absence of (other) linear features (1)
-	-	Small number (3)	Unvegetated fences and large field sizes (2)
Rarer (5)	Small number of bats (10)	Moderate number/Not known (4)	Walls, gappy or flailed hedgerows, isolated well-grown hedgerows, and moderate field sizes (3)
-	-	Large number of roosts, or close to a SSSI for the species (5)	Well-grown and well-connected hedgerows, small field sizes (4)
Rarest (20)	Large number of bats (20)	Close to or within a SAC for the species (20)	Complex network of mature well-established hedgerows, small fields and rivers/streams (5)

Table 5: Valuing foraging areas

Species	Number of bats	Roosts/potential roosts nearby	Foraging habitat characteristics
Common (2)	Individual bats (5)	None (1)	Industrial or other site without established vegetation (1)
-	-	Small number (3)	Surburban areas or intensive arable land (2)
Rarer (5)	Small number of bats (10)	Moderate number/Not known (4)	Isolated woodland patches, less intensive arable and/or small towns and villages (3)
-	-	Large number of roosts, or close to a SSSI for the species (5)	Larger or connected woodland blocks, mixed agriculture, and small villages/hamlets (4)
Rarest (20)	Large number of bats (20)	Close to or within a SAC for the species (20)	Mosaic of pasture, woodlands and wetland areas (5)

In our experience, using the approach outlined above for real situations we have been involved with generally produces a result which corresponds with our professional judgement on valuation of the receptor. But it would be counter-productive for ecologists to use this approach without a critical eye on the result produced: if there are factors not covered above that make a particular receptor valuable at a higher, or lower, geographic frame of reference. This is entirely appropriate if it can be adequately justified.

Consequently, this approach should be used as a framework rather than a rulebook, but one which could help all of us involved in preparing EcIAs to be more consistent and to better explain our rationale in the valuation of bats as receptors. We are aware that many ecologists have asked us for copies of this framework following our presentation in 2007, and we would be very interested to hear how well you have found it to work in practice, and any suggestions for improvement.

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Photo 1. Gas pipeline severing bat commuting routes



Photo 2. Lesser horseshoe bat Photo: David Wells



Photo 3. Example of bat commuting habitat in an intensively managed landscape



Photo 4. Example of bat foraging habitat in a 'patchy' landscape Photos 1,3,4: Cresswell Associates

Exploring behavioural responses of shorebirds to impulsive noise

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Abstract

Despite recent research interest in the consequences of chronic noise for animal populations, the effects of impulsive noise are less well understood. Determining wildlife responses to impulsive sound is pertinent in coastal areas where development of port and power generation industries may result in disturbance from impulsive sounds such as percussive piling, especially around estuaries which may support internationally important numbers of bird species. Discussions between regulators, planning authorities and regional development agencies over precautionary levels of impulsive sound emitting activities are common, yet we're unaware of any study which provides guidance on acceptable noise limits that would keep disturbance to shorebirds to a minimum. This experimental study intentionally disturbed birds at a high tide roost site, an agricultural field adjacent to the sea wall on the south bank of the Humber estuary in northern England, using an impulsive sound, and their behavioural responses were recorded. The researcher sounded an air-horn at ever decreasing distances towards a mixed species flock of shorebirds. While visual disturbance from the experimenter was taken into consideration in the methods, we cannot statistically separate their effects in this study. An ordinal logistic regression model demonstrated a statistically significant positive relationship between the decibel (dB(A)) level experienced by the birds and the behavioural response observed. In principle, this model can be used to predict the probability of a particular behavioural response at a given dB(A) level. Estimates of sound levels which included calculation of geometric and atmospheric attenuation were superior over more simple measures in predicting behavioural responses. This study has implications for assessing possible disturbance caused by impulsive noise, and adds

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to growing evidence that the effects of noise may be an important, albeit not straightforward, consideration in management decisions made for shorebirds and other avian species.

Key words: coastal development, disturbance, human-wildlife conflict, Humber Estuary, impulsive noise, waders.

There has been increasing concern raised over the consequences of human-induced noise to animal populations and the integrity of protected areas (Brumm 2004; Brumm & Slabbekorn 2005; Habib *et al.* 2007; Barber *et al.* 2010). Determining the severity of the impacts of disturbance from human activity on wildlife is essential for appropriate management decisions (Gill 2007), yet quantifying these has proven challenging. Risk assessment for potentially disturbing human activity (*e.g.* construction) is based on the precautionary principle, and the absence of scientific knowledge does not necessarily result in postponement of mitigation (Myers 1993). There is generally an assumption that noise disturbance is harmful to birds and that there is a safe threshold noise level at which noise does not have a significant effect on birds using a site; moreover, this may be extrapolated as being constant across all species and sites. Yet given differences in background noise levels and the potential for birds to habituate to some types of disturbance, the response to auditory disturbance (and thus the maximum noise level at which a behavioural response is seen to be absent) is likely to vary quite substantially between species, across sites and seasons, and for different types and levels of sound.

There has been recent research into the effects of noise exposure on wildlife. Direct effects of chronic noise exposure include hearing loss, increased production of stress hormones and hypertension (Barber *et al.* 2010). Indirect effects include the masking of acoustic signals such as calls and sounds which may otherwise lead to alert behaviour (Slabbekorn & Ripmeester 2008), as well as the reduced perception of unintentional sounds which may attract predators (Barber *et al.* 2010). It has also been suggested that exposure to chronic noise may result in increased vigilance in order to compensate for decreased auditory awareness (Quinn *et al.* 2006). All of these effects of chronic noise have clear deleterious consequences on resident or breeding animals at an individual level, and reductions in population density with proximity to noise sources are also well documented (Reijnen *et al.* 1995; Forman & Alexander 1998), as well as possible effects on demography (Habib *et al.* 2007). Minimising the effect of noise from human activity on wildlife using protected areas therefore is important for the management of these sites. Yet most conservation issues relating to noise in coastal areas concern impulsive noise disturbing wintering or migrating populations during the short-lived construction phase of a development, where the consequences are less well known.

Impulsive noise may be defined as a short duration noise extending over a range of

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frequencies. The effects of impulsive noise on wildlife behaviour are pertinent to coastal areas experiencing intensive development due to the requirement of ground stabilising construction methods such as percussive piling. This process produces periodic impulsive noise of high intensity which has led to concerns over changes in distribution and abundance of coastal birds due to disturbance (Cutts *et al.* 2008). At present there is little evidence for how impulsive noise relates to associated behavioural responses, such as dispersal, movement or site abandonment. Regulation of impulsive noise at or adjacent to protected areas therefore is based upon previous, largely non-noise based, disturbance research, and needs to be verified.

The effects of impulsive noise may also differ between species due to differing energy requirements and food sources. For instance, Common Redshank *Tringa totanus* are regarded as being more vulnerable to the effects of disturbance and extreme weather than other waders, due to their requiring more feeding time to offset the energetic costs of foraging (Mitchell *et al.* 2000). Prevailing geographical and environmental conditions also affect behavioural responses and thus the effects of disturbance; birds are more likely to respond if there is suitable habitat present nearby to which to retreat (Gill *et al.* 2001). Likewise, in the case of impulsive noise, propagation of a potentially disturbing sound is highly dependent upon meteorological conditions. So, for example, there may be occasions when the effects of impulsive sound are more likely to result in site abandonment because they are emitted upwind of a roost site and the nearest available alternative roost is downwind. The wide range of environmental and physiological factors affecting species-specific and site-specific costs in responding to disturbance increases the likelihood that a fixed cap on sound emissions, to minimise disturbance to shorebirds and other species, may fail to meet its objectives.

This project aims to develop a probabilistic approach for assessing the consequences of impulsive noise thresholds for roosting shorebirds. It explores the behaviour of multi-species shorebird flocks (comprised of Common Gull *Larus canus*, Curlew *Numenius arquata*, Golden Plover *Pluvialis apricaria* and Lapwing *Vanellus vanellus*) in response to varying sound levels, and determines the probability that a given level of impulsive noise will lead to one of an increasing range of behaviours, from no observed effect to the birds flying from the study area. It is hypothesised that increased impulsive noise will lead to more energetically expensive alarm or fleeing behaviour. This approach should provide government conservation agencies with better information for regulating impulsive noise thresholds for different shorebird species or sites (*i.e.* for minimising industrial and recreational disturbance levels), should be acceptable to stakeholders and also contribute to the growing literature on this important issue for wildlife management.

Materials and methods

Study site and species information

Great Britain is internationally important for wintering waders and wildfowl due to its

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extensive estuarine habitats and mild climate, and the Humber Estuary has been estimated to account for around 170,000 of these birds, including internationally important numbers of several species (English Nature 2004; Musgrove *et al.* 2007). The estuary is designated as a Special Protection Area (SPA) under the Birds Directive (EC Directive on the conservation of wild birds; 79/409/EEC), as a Special Area of Conservation (SAC) under the Habitats Directive (Council Directive 92/43/EEC of 21 May 1992) and qualifies as a Wetland of International Importance under the Ramsar Convention. The estuary is also a busy commercial waterway and the south bank of the estuary includes some of the most industrialised parts of England, comprised of chemical works, oil refineries and power stations (Gameson 1982; English Nature 2004). Development of regional industry is important for employment and the local economy, but its proximity to the Humber Estuary SPA gives rise to concern over the potential consequences of chronic and impulsive noise from industry for birds on SPA.

Given the protection accorded to a number of coastal sites around Britain, permission may be required for any plan or project to be conducted on or near them (Natural England, pers. comm.), including experimental research. The study described here, which included noise disturbance of wintering shorebirds, was granted permission but required that our activity would have a negligible effect on the long-term protected status of adjacent sectors of the Humber Estuary SPA. Due to the difficulty in gaining both regulator (Natural England) and owner permission for multiple sites, experimental disturbance was carried out from January–March 2010 on a single site close to Stallingborough, northeast Lincolnshire (53°35'N, 0°08'W). The status of the site (which is currently agricultural but earmarked for development) facilitated permission being obtained for the study near Stallingborough. Permission was also granted on the basis that the study was being made at the end of the winter, when flock sizes were smaller, and our agreeing to conduct disturbance events only when the weather was mild (*i.e.* in the absence of strong winds, and when temperatures were above zero; see below).

The study site consisted of a large farmland field, close to a number of industrial power plants, used by several species of shorebirds as a high tide roost. Over the study period the area was used by small flocks (range = 4–34 individuals, median = 14) of Common Gull, Curlew, Golden Plover and Lapwing. While these numbers were as expected for the time of year, this site and the surrounding area attracts large numbers of waterbirds during migration and winter; for example, a peak November count of > 4,000 Golden Plover was recorded in 1999 (Austin *et al.* 2008).

Sound monitoring and behavioural observations

Two Casella CEL-593 sound level analysers were used for monitoring both ambient and impulsive noise levels, using a 1/3 octave band filter to measure the energy average L_{Aeq} (defined as the equivalent continuous sound which would contain the same sound

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energy as the time-varying sound over the same time period). Use of 1/3 octave band measurement provides better resolution, in terms of the frequency components of a given sound, than broad band or octave band; on the CEL-593 this constitutes a range of 33 bands ranging from 12–20,000 hz as defined by the International Standards Organisation (ISO) (ISO 266:1997), each with an associated decibel level. The CEL-593 also includes an event mode whereby any sound detected over a particular threshold level will have its length and maximum level recorded separately from the background measurement in progress. A threshold level of 71dB(A) was found to be appropriate for detecting most intentional disturbances whilst preventing false detection of variation in background noise levels or other incidental noise such as gusts of wind. The high threshold was required due to the close proximity of operational power production plants and associated traffic. A-weighting of decibels was used when calculating overall broadband sound levels. This weighting scheme, which is based on the inverse of a curve of sound intensities perceived as being equally loud to humans, has also been found to approximate to hearing in birds (Dooling & Popper 2007).

Fieldwork was carried out on days forecasted for fair weather and winds of < 5 m/s in order to ensure accuracy of sound monitoring. On four visits where wind speed was > 5 m/s fieldwork was abandoned because, despite wind shields being used, extraneous noise is created as higher speed winds move over the microphone. Six days of sampling were completed in 2010, on 30 January, 20, 25 and 27 February, and 7 and 12 March 2010. The instruments were set up at least 2 h prior to high tide while birds were feeding on the estuary, in order to avoid unintentional disturbance. They were placed in the centre of the field at 165 m from either end, at a height of 1 m, in order to provide maximum coverage for possible locations of roosting birds (Fig. 1). Observers then left the field and waited for high tide. The instruments were left running to provide an average

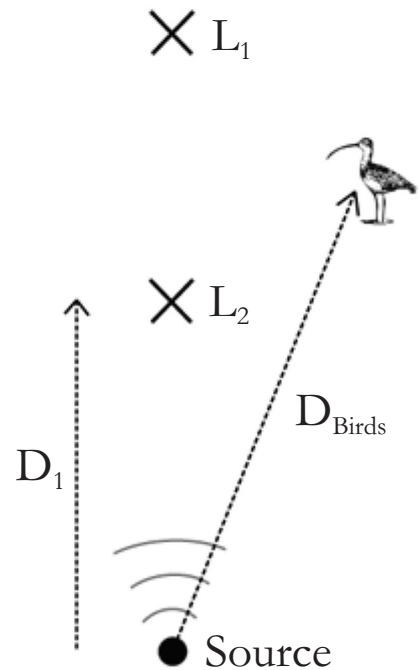


Figure 1. Placement of sound recording instruments (X L_1 and X L_2) relative to the source of the sound. By using sound levels recorded at a known distance (*e.g.* D_1) to calculate the power of the source it is possible to determine approximately the decibel (dB(A)) level experienced at other distances (D_{Birds}).

ambient noise level (L_{Aeq}) estimate for the measurement period (approximately 1–2 h), and because it was not possible to manipulate the equipment once the birds had arrived without causing unintentional disturbance. Shortly after high tide the number of birds present on the site was counted and their position relative to the two instruments (measured from the first location by compass) was recorded, for use in estimating the decibel level they experienced. The birds show great fidelity to their roosting location within the field and the maximum angle between the observer, the downrange meters and the birds was 35° (see Appendix A).

Starting at the end of the field furthest from the majority of birds present (typically around 250 m) an air horn was used as a sound source to cause intentional disturbance (FPS air horn, 114dB(A) measured at 2 m from source; www.screwfix.com). The air horn was sounded for a standardised 3 s per replicate. Responses by the birds were classed as follows:

0 – No behavioural changes observed
1 – Behavioural change (*e.g.* vigilance or alarm calls) but not flight
2 – Flew but soon returned to the site
3 – Flew and abandoned the site.

The number of birds showing each type of behavioural response within the flock was also recorded. If there was no observable behavioural change in any of the birds (or else after a non-flight response had ended) the distance to the instruments was decreased by 20 m, and the horn was sounded again. If the sound resulted in a flight response then we would wait until the birds returned, or could be confirmed as having abandoned the site before proceeding. Birds that flew out of site were considered to have abandoned the study area. Five minutes were allowed to pass between replicates, starting from when birds were settled to reduce the effects of visual disturbance (*i.e.* the birds seeing the researcher), which is also likely to affect the behaviour of the birds (Rees *et al.* 2005; Bregnballe *et al.* 2009). Once all birds had abandoned the site the instruments were turned off and removed. Sound data were downloaded using the software dB5 (Casella USA), and exported to an Excel spreadsheet containing the time, date, distance to each instrument, species, flock size and the birds' response to each intentional disturbance event. The distance from the source to each flock was determined, based on their position relative to the two instruments.

Downrange noise and atmospheric attenuation

In order to obtain an approximate measure of the decibel level of the intentional disturbance experienced by a given flock of birds, an estimate of the attenuation of the sound over distance due to geometrical divergence (spherical spreading in the free field from a point sound source, based on the distance from the source to the receiver, A_{div}) and atmospheric absorption (attenuation coefficient dependent upon temperature, pressure and humidity, A_{atm}) was calculated. A power level (L_W) for the source was calculated based on the readings at each instrument, which was then used to determine the approximate decibel level received by the birds at a given distance

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(L_{Birds}). Full details of the calculations and our justification for using a spherical spreading model of attenuation in the present study are given in Appendix A.

Statistical analysis

Ordinal logistic regression analysis was carried out in R 2.11.1 using the package Design 2.3-0 (R Development Core Team 2010). Behavioural response was ranked by severity (0–3) to be used as an ordinal dependant variable. A maximal model (MAM) including all possible factors and covariates (Response ~ L_{Birds} + Species + Flock size + Temperature + Pressure + Humidity) was reduced sequentially by dropping non-significant terms. No interactions were included due to low sample sizes. The MAM could then be used to make predictions of the probability of a particular behavioural response at a given level of L_{Birds}. The proportional odds assumption, which states that the coefficient between pairs of outcome groups remains constant (*i.e.* the coefficient between response 0 and 1 is equal to that of 1 and 2), was tested using a graphical approach suggested by Harrell (2001) and was found to hold. A chi-squared goodness of fit was performed to test the fit of the model to the original data.

As birds disturbed at the start of the study were potentially the same birds disturbed on subsequent days, and the same birds were also likely to be consecutively disturbed within days, there was a high probability of pseudoreplication in the dataset. The ordinal logistic package used to predict probabilities of multiple behaviour types over the scaled impulsive noise levels was not able to account either for repeated measures or for random effects to control for this. Generalized linear mixed models (lmer from R package "lme4" 0.999375-35) and generalized estimating equations (geeglm from R package "geepack" 1.0–17) were used to examine whether the relationship between L_{Birds}, species and behavioural response would remain when including random effects to account for pseudoreplication (see Appendix B). While the effect of L_{Birds} was reduced on taking into account repeated measures made across days, it remained positive and statistically significant. The ordinal logistic regression results are therefore provided on the basis that this pseudoreplication does not appear to detract significantly from the qualitative conclusions of this study, while allowing presentation of the results in a probabilistic format (see later). Methods to combine the ordinal regression with repeated measures and random effects are under development (see Appendix B).

Linear discriminant analysis (R package "MASS" 7.3–7) was conducted as a *post-priori* assessment of the possibility of increasing the availability or complexity of information useful for predicting bird behaviour. This would be of value for determining the most suitable methods for obtaining accurate estimates of local behavioural responses to impulsive noise disturbance. Behavioural response was the categorical dependent variable while species and various measures of L were used as factors. The rate of grouping accuracy (%) was calculated by comparing predicted and observed responses for each case. By using different measures of L in separate analyses

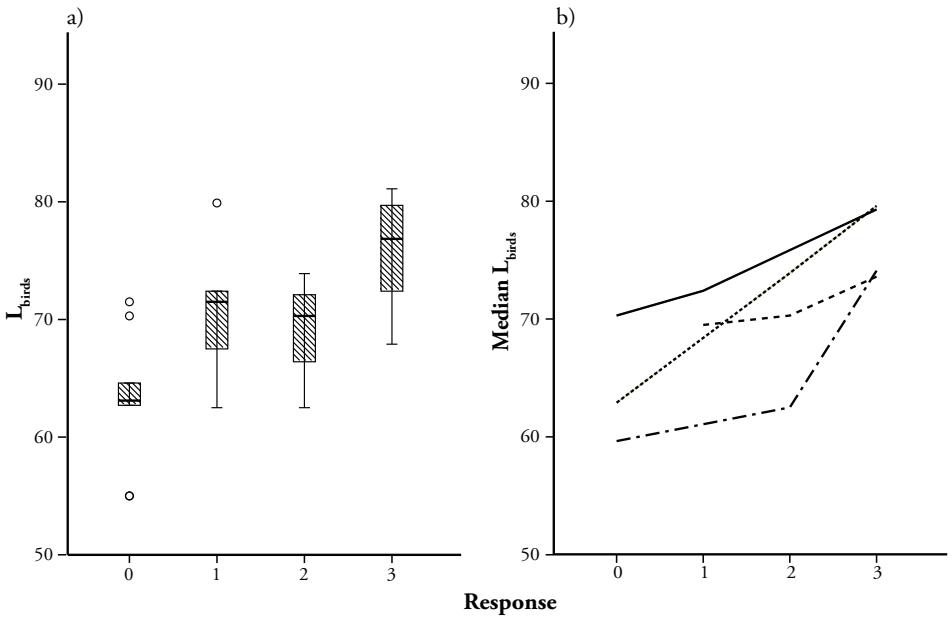


Figure 2. a) Boxplot showing the relationship between behavioural response and dB level experienced by intentionally disturbed birds. The line in the centre of the box represents a median value, and the top and bottom represent 25th and 75th percentiles. Whiskers represent minimum and maximum values excluding points that fell outside 95% confidence intervals. b) The relationship between behavioural response and median dB level experienced by four different species of bird which were intentionally disturbed. Full line = Curlew, dotted line = Golden Plover, dashed line = Common Gull, Dot-dash = Lapwing.

it was possible to analyse the reliability of using the recorded sound level at a single instrument (L_1 and L_2) or arithmetic averages between two microphones (L_{1+2}) in prediction of behavioural response.

Results

Relationship between L_{Birds} and behavioural response

The relationship between L_{Birds} and behavioural response is shown in Fig. 2a. L_{Birds} was highly variable for no observable behavioural response (median = 63.1 dB(A), range = 54.9–71.5 dB(A) with a high proportion of extreme outliers. Flight with all birds abandoning the site (median = 76.85 dB(A), range = 67.9–81.1 dB(A)) appears to occur at notably higher noise levels than any other response, though its lowest value does fall below the median of the other two behavioural responses. If separated by species it is clear that flight with abandonment is indeed seen at higher levels, though there is inter-specific variation in the thresholds observed (Fig. 2b). The noise levels that elicit non-flight behavioural (median = 71.5 dB(A), range = 62.4–79.1 dB(A)) and flight with return (median = 70.3 dB(A), range = 62.4–73.9 dB(A))

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responses appear to show similar values on grouping species (Fig. 2a), but not when the species are considered separately (Fig. 2b). In addition to the apparent difference in thresholds between species, there is also a notable difference in the slope of the association; for example, the slope for the Common Gull is notably less than that observed for Golden Plover. In terms of its behavioural response, Lapwing appears to be the species most sensitive to intentional disturbance, while Curlew is the most tolerant (Fig. 2b). There was no detectable effect of sensory loading (*e.g.* a decrease in thresholds due to repeated treatments), but see Appendix B.

Ordinal logistic regression

There was a significant relationship between dB(A) level experienced (L_{Birds}) and behavioural response (Wald Z = 3.58, Coefficient = 0.277, $P < 0.001$). All other factors were found to be non-significant. Testing for an interaction between L_{Birds} and species, which preliminary data exploration suggested may be useful in describing variation in response (Fig. 2b), was not possible in the present study due to low sample sizes. In a model containing L_{Birds} and species as factors, the latter was found to be non-significant (Wald Z = 1.60, Coefficient = 0.776, P = 0.11, n.s.). Goodness of fit tests indicate that the MAM does not differ significantly from the observed values for behavioural response (χ^2_{66} = 68.82, $P > 0.38$, n.s.). The ordinal logistic regression model was also used to predict probabilities of behavioural responses for given levels of L_{Birds} (Fig. 3). Intentional disturbance at very low dB(A) levels is highly unlikely to elicit a behavioural response, while at above 65.5 dB(A) a behavioural response of some kind becomes more likely to occur than no response. At above 72.2 dB(A) flight with abandonment of the site becomes the most likely outcome of the disturbance. If non-response and non-flight response were taken to be relatively harmless, and flight responses potentially costly (in terms of energy expenditure), then for those species studied at the site a costly outcome becomes more likely at = 69.9 dB(A).

Linear discriminant analysis

The accuracy of the prediction of observed values differed by the measure of dB(A) used, as shown in Table 1. L_{Birds} which is the measure including attenuation calculations, was the most accurate at predicting the correct response (64.5%). Readings at single instruments without calculations (L_1 and L_2) were less accurate than this (38.7% and 54.8%, respectively), as was using a mean L value of the two instruments (51.6%). The predictions given by using L_{Birds} were within a single measure of the response (*e.g.* predicting a type 0 or type 1 response for cases where no behavioural response was observed) for the vast majority of cases (93.6%), and were almost as successful at predicting harmless against harmful responses (as defined, Fig. 3) 87.1% of the time.

Discussion

Noise and behaviour

All bird species subject to intentional disturbance showed a range of behavioural responses, with the probability of a more

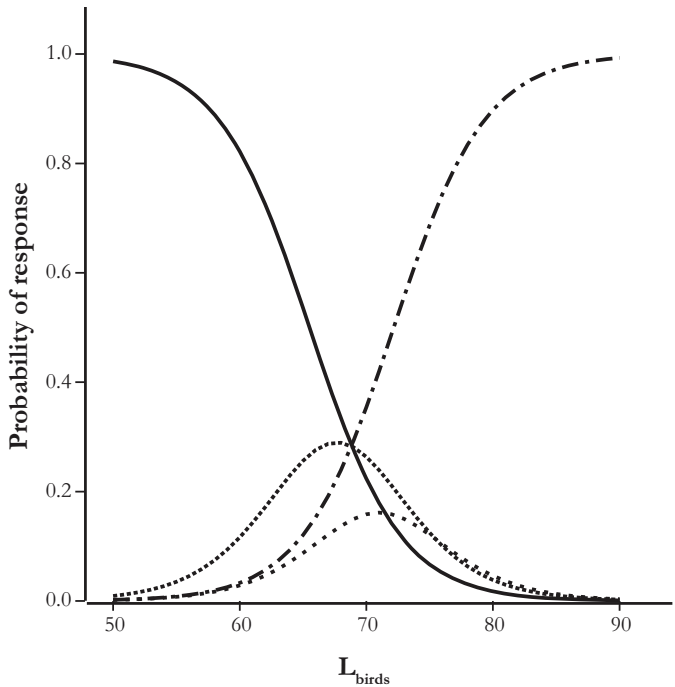


Figure 3. Probability of behavioural response based on dB(A) level experienced based on predictions from an ordinal logistic regression model. Solid line = no behavioural response observed (0), dotted line = behavioural change, but did not fly (1), dashed line = flew but returned to area (2), dot-dashed line = flew and abandoned the area (3).

Table 1. Accuracy of predictions of behavioural response to intentional disturbance for four different measures of decibel level using a linear discriminant analysis.

Measure of decibel level	**% Prediction accuracy**		
	Exact	**Within 1 response level**	**Non-flight *versus* flight response**
L_1	38.7%	61.3%	45.2%
L_2	54.8%	87.1%	67.7%
L_{Mean}	51.6%	90.3%	71.0%
L_{Birds}	64.5%	93.6%	87.1%

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energetically costly behavioural response occurring increasing with noise levels experienced. The severity of the response observed also increased with noise levels, in a species-specific manner. If we assume that behavioural responses that include flight are energetically costly, could potentially affect the survival or productivity of an individual bird and thus contribute to negative trends in bird numbers locally, then impulsive noise limits should be restricted to < 69.9dB at this site (but see discussion of caveats below). There was no significant difference in the behavioural responses of the four shorebird species considered in this study in response to impulsive noise, but this may be attributable to the low sample size.

The behavioural response of birds to human activity is thought to depend upon associated predation risk and energetic cost (Yasue 2005), with humans perceived as predators (Frid & Dill 2002), yet birds are known to respond to impulsive noise without visual stimuli. As remote noise-making equipment was not used in this project, it is likely that the increase in severity of behavioural responses with noise levels was to some extent confounded by visual stimuli. D_{Birds} (the distance from the sound source to each flock) was non-significant in the ordinal logistic regression model, but when used as a predictor in the linear discriminant analysis it was similar to a single instrument at predicting behavioural responses. At face value this suggests that behavioural responses largely depended upon the visual stimulus of the researcher approaching the birds. However, this relationship is more likely to be due to D_{Birds} showing a relatively strong negative correlation with L_{Birds} (the noise level experienced by the birds). This is unsurprising as the calculated power level (L_W) of the source was approximately the same for each repeat, and attenuation increases with distance. In a practical situation where impulsive noise events may vary considerably in both power and frequency components D_{Birds} is likely to be a poor predictor of behavioural response. Nevertheless, there is likely to be a threshold distance where L_{Birds} is low enough as not to be perceived against background noise, but determining this accurately in practice would require knowledge of the likely sound sources, the frequency components they produce, and atmospheric conditions. Behavioural responses resulting from walking towards the flock, prior to the use of the air horn were rare. Where birds responded we paused until they had returned and settled or until they had left the area before continuing with the experimental noise with remaining birds. Nevertheless, it is likely that a combination of visual and auditory disturbance stimuli resulted in the responses observed, particularly in this study area where chronic ambient noise exposure (average L_{eq} = 68dB(A)) may have resulted in increased vigilance (Quinn *et al.* 2006). This combined effect may have resulted in increase in the severity of responses observed; use of remote sound-making equipment and also testing the effects of approaching the birds without using noise (*i.e.* quantifying the visual factor) will help to clarify this issue.

Behavioural responses to intentional noise disturbance appear to differ in a

species-specific manner, though an increased sample size will be required to determine whether this effect is biologically relevant. Blumstein *et al.* (2005) suggested that body size be used as a predictor of responses to disturbance, with body mass positively correlating with the distance at which birds fly from a predator. Contrary to this hypothesis, the largest species in this study, Curlew, appeared to be the species most tolerant to noise disturbance, while Lapwing showed the lowest apparent thresholds. This may be due to differences in hearing, or to differences in perceived predation risk posed by impulsive noise, as opposed to approaching pedestrians. Alternatively, there may be a difference in the amount of feeding opportunities required to compensate for the flight response relative to the time of the tide, the distance to the nearest alternative roost in this linear landscape and the costs associated with flight for species of different size (Mitchell *et al.* 2000). Common Gulls had relatively low noise thresholds, with the least steep slope between non-flight and flight response. When abandoning the site at high tide, Common Gulls were able to retreat to the adjacent estuary as they are able to swim on the water, while waders would have to fly to a different inland roost site as their feeding habitat is not accessible at high tide. This may represent a difference in energetic costs affecting inter-specific responses to disturbance, though it will be necessary to test this assumption at different sites with different associated costs.

The study was made at a time of winter when the flock sizes in the study area were small and relatively invariable in order to gain the permissions needed to undertake work. This may have affected the estimates of noise thresholds; although flock size was not found to be significant in predicting behavioural responses in this analysis, other studies indicate that it influences foraging efficiency and flight distances in response to human disturbance in waterbirds (Mori *et al.* 2001; Yasue 2005).

Practical considerations and applications

Despite the caveats regarding confounding variables, some recommendations and guidelines can be provided to developers, government conservation agencies and non-governmental conservation organisations. The study set out to explore whether simple behavioural observations can provide a probabilistic framework of behavioural responses of shorebirds to potentially disturbing elements of the human environment. We have, for a given site, shown that the likelihood of eliciting a particular alarm or flee behaviour in roosting shorebirds can be determined by estimating noise thresholds that take into account distance between the noise emission and the roost and atmospheric conditions, *i.e.* what the birds actually hear down-range of a sound source. Our discussions with those involved in the development and conservation of estuarine environments suggest that this site-based approach would be preferable to adopting a precautionary principle across sites. We therefore recommend that research-led estimates of the risks of noise disturbance to shorebirds be put into practice more widely.

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It is important that the effects of impulsive noise on wildlife be addressed, particularly in areas undergoing extensive development. Deleterious effects of chronic noise exposure have been suggested to begin at levels as low as 55–60 dB(A) (Dooling & Popper 2007), though data on physiological effects are lacking. The effects of chronic or impulsive noise on the population density of wintering birds is also poorly understood, yet should be addressed if development of modern energy sectors around the UK coastline is to be harmonised with the UK's commitment to protecting internationally important wetland sites.

It is unclear whether it is the perceived change in impulsive noise in relation to ambient noise, or simply the level of the impulsive sound itself which causes the behavioural responses. This is important in how limits are defined and comparison studies with quieter sites will be necessary to determine which of these best represents birds' recognition of impulsive sound. Nevertheless the relationship between predicted probability of a behavioural response and the estimated sound levels resulting in a behavioural response (L_{Birds}) provides a reasonable best practice method for regulators and regional authorities to define noise limits based on a likelihood of risk, for a given site based on its ambient noise level. Of course, even if another site has a similar level of ambient noise (L_{eq}), the host of environmental and physiological factors affecting behavioural responses will no doubt change the relationship, but L_{eq} is relatively easy to measure in practice.

The linear discriminant analysis indicates that L_{Birds}, which takes account of noise attenuation, is superior to simple measures of dB(A) level by single instruments at predicting behavioural responses (Table 1). We are not the first study to draw attention to the need to quantify the "dose" or received noise at the position of the animal for studies of anthropogenic disturbance to be meaningful (Nowacek *et al.* 2001; Barber *et al.* 2010). While it would not be possible to determine the precise dB(A) experienced by flocks relative to instruments during monitoring activity, this may improve accuracy if likely distances, sounds and atmospheric conditions were considered prior to monitoring.

Conclusions

This study represents a first step towards addressing the ecological, environmental and regulatory issues affecting how noise limits are defined. The role of habituation, differences between species responses to different sounds, seasonal differences, energetic effects, carry-over effects and the relationship between ambient and impulsive noise levels should be assessed in a systematic fashion in order to provide a full understanding of the effects of human induced noise on wintering shorebirds. However, where the aim is to provide meaningful limits that minimise disturbance effects of coastal development on wildlife and to reduce precautionary limits where conflicts between industrial, regulatory and conservation stakeholders often arise, experimental approaches such as this one are useful in controlling for some of the vast amount of confounding factors influencing disturbance, its effects and its consequences.

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Appendix A: Calculation of impulsive noise disturbance experienced by birds, taking account of atmospheric attenuation and directionality.

This Appendix describes the methods used to calculate the L_{Birds} level from the instrument recordings.

Attenuation calculation

In order to determine the decibel level of the intentional disturbance experienced by a given flock, an estimate of the attenuation of the sound over distance due to geometrical divergence (A_{div}) and atmospheric absorption (A_{atm}) had to be calculated. An approximate sound power level (L_W) for the source was calculated based on the readings at each instrument, which could then be used to determine the decibel level at a given distance (L_{Birds}) by adding the two attenuation coefficients:

$$L_{Birds} = L_W + A_{div} + A_{atm}$$

The geometrical divergence accounts for spherical spreading in the free-field from a point sound source, based solely on the distance (*d*, in metres) from the source to the receiver:

$$A_{div} = \left[-20 \log_{10} \left(\frac{d}{d_0}\right) + 11\right]$$

It is recognised that the assumption of spherical spreading (*i.e.* assuming that sound energy radiates from the source in an equivalent way in all directions) when the air horn is a directional sound source, and thus the omission of directionality in the divergence calculation for L_{Birds}, is a limitation of the methods used in this study which will be addressed in future studies. Meanwhile, the significance of omitting directionality on this occasion was considered to be reduced by both the sound level meters and the birds being positioned within a (maximum) 35° cone in front of the sound source.

The attenuation due to atmospheric absorption is given by:

$$A_{atm} = \frac{\alpha d}{1000}$$

where α is the atmospheric attenuation coefficient in decibels per km and d is distance in metres as above. This was calculated using an online calculator provided by the National Physical Laboratory (http://resource.npl.co.uk/acoustics/techguides/absorption/), based on equations described in ISO 9613-1. Measurements of temperature, pressure and % humidity were obtained for each day from a private weather station at Grimsby.

As atmospheric attenuation varies in relation to the frequency of the sound in question, L_W values had to be calculated for the source for each 1/3 octave band recorded in order to give a meaningful power level:

$$L_{W1} = 10 \log_{10} \left[\sum_{j}^{33} 10^{0.1[L_1 - A_{div} - A_{atms}]}\right]$$

where j is an index indicating thirty-three 1/3 octave band frequencies and L_1 is the decibel level for a single band observed at the first instrument. It was necessary to log transform and then back-transform when summing decibels as they are logarithmic units. This calculation was also carried out for the second instrument to give a second estimate of the power level (*i.e.* L_{W2}).

Values for L_{W1} and L_{W2} were then compared; in ideal conditions these would be identical as they are both estimates for the same source, but variables not accounted for in the calculations (*e.g.* reflections from the ground and the effects of wind) lead to variation. Where these differed by > 5dB the L_W for the closest instrument to the source was used, as this was likely to be less affected by other factors. If the two L_W estimates were within this range the mean value was used. These estimates could then be used to calculate the decibel level experienced by the birds, as shown in the first equation.

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Appendix B. Accounting for Pseudoreplication.

As sampling occurred at the same site on multiple days, and birds were exposed to multiple noise replicates within those days, it is possible that the variance in behavioural responses observed were due to either, or both, of these factors. This pseudoreplication is not accounted for in the minimum adequate model described in the results section (for reasons described below); however, the analyses below suggest that the positive relationship between L_{Birds} and behavioural response remains significant even when accounting for these factors.

The results of generalized linear mixed models (GLMM) using the package "lme4" in R are shown in Tables B1 and B2. In the first, noise replicate was specified as a random term; in the second, day was specified as a random term, with noise replicate as a nested term within day. As with the results of the main analysis, species was found to be non-significant (Table B3). The results of a generalized estimating equation model (GEE) using the package "gee" are shown in Table B4. Noise replicates was included as a random term.

Table B1. Relationship between L_{Birds} and behavioural response from generalized linear mixed model including noise replicates as a random term.

	Estimate (s.e.)	**Z value**	***P***
Intercept	–22.02 (8.63)	–2.55	0.011
L_{Birds}	0.34 (0.13)	2.62	0.009

Table B2. Relationship between L_{Birds} and behavioural response from generalized linear mixed model including day and noise replicates as random terms.

	Estimate (s.e.)	**Z value**	***P***
Intercept	–33.83 (13.00)	–2.60	0.009
L_{Birds}	0.51 (0.19)	2.66	0.008

Table B3. Maximal model of the GLMM shown in Table B2.

	Estimate (s.e.)	**Z value**	***P***
Intercept	–63.40 (24.93)	–2.54	0.011
L_{Birds}	0.93 (0.35)	2.63	0.009
Species	0.93 (1.01)	0.92	0.35 (n.s.)

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Table B4. Relationship between L_{Birds} and behavioural response from generalized estimating equation model.

	Estimate (s.e.)	**Wald**	***P***
Intercept	–6.71 (1.47)	20.98	<0.001
L_{Birds}	0.13 (0.02)	49.81	<0.001
Day 2	–0.13 (0.55)	0.06	0.81 (n.s.)
Day 3	–0.67 (0.63)	1.13	0.29 (n.s.)
Day 4	0.38 (0.56)	0.45	0.50 (n.s.)
Day 5	0.26 (0.54)	0.23	0.63 (n.s.)
Day 6	–0.07(0.54)	0.02	0.90 (n.s.)

While use of statistical approaches such as GLMM and GEE allow us to take account of the apparent pseudoreplication in the data, they do not simultaneously provide predicted probabilities for each potential behavioural responses for a given value, as outlined in the main text for multiple ordinal logistic regression. This feature was central to the idea of a risk-based analysis relevant to impact assessments, and while the model outputs should be interpreted with care, we feel it was justified to take this approach to statistical analysis. For future studies with similar requirements, i.e. repeated measures in a proportional odds logistic regression to allow for random effects and serial auto-correlation (see R package "repolr"), may represent an appropriate method if it is developed to allow predictions across all ordinal responses simultaneously (Parsons *et al.* 2009).

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