From:	Gemma Keenan
To:	Tracey Williams
Cc:	[redacted]
Subject:	Norfolk Vanguard - Email 5 of 18 Deadline 1 Submissions
Date:	16 January 2019 14:55:14
Attachments:	ExA;WQApp03.05;10.D1.3 Norfolk Vanguard WQ Appendix 3.5 Furness 2015.pdf ExA;WQApp03.06;10.D1.3 Norfolk Vanguard WQ Appendix 3.6 Wade et al. 2016.pdf ExA;WQApp03.07;10.D1.3 Norfolk Vanguard WQ Appendix 3.7 Furness et al 2018.pdf ExA;WQApp03.08;10.D1.3 Norfolk Vanguard WQ Appendix 3.8 Cleasby et al. 2015.pdf ExA;WQApp03.09;10.D1.3 Norfolk Vanguard WQ Appendix 3.9 Garthe and Hüppop 2004.pdf ExA;WQApp03.10;10.D1.3 Norfolk Vanguard WQ Appendix 3.10 Green et al 2016.pdf ExA;WQApp03.11;10.D1.3 Norfolk Vanguard WQ Appendix 3.11 McGregor et al 2018.pdf

Dear Tracey

This is email 5 of 18 of the Applicant's submission for Norfolk Vanguard Examination Deadline 1.

We enclose the following documents:

Appendices to Written Questions:

- Appendix 3.5 Furness, R.W. (2015). Non-breeding season populations of seabirds in UK waters: Population sizes for Biologically Defined Minimum Population Scales (BDMPS). Natural England Commissioned Reports, Number 164.
- Appendix 3.6 Wade, H.M., Masden E.M., Jackson, A.C. & Furness, R.W. (2016). Incorporating data uncertainty when estimating potential vulnerability of Scottish seabirds to marine renewable energy developments. Marine Policy 70:108-113
- Appendix 3.7 Furness, R.W., Garthe, S., Trinder, M., Matthiopoulos, J., Wanless, S. & Jeglinski, J. (2018) Nocturnal flight activity of northern gannets Morus bassanus and implications for modelling collision risk at offshore wind farms. Environmental Impact Assessment Review, 73, 1-6
- Appendix 3.8 Cleasby, I.R., Wakefield, E.D., Bearhop, S., Bodey, T.W., Votier, S.C. & Hamer, K.C. (2015) Three-dimensional tracking of a wide-ranging marine predator: flight heights and vulnerability to offshore wind farms. Journal of Applied Ecology, doi: 10.1111/1365-2664.12529
- Appendix 3.9 Garthe, S and Hüppop, O. (2004). Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index. Journal of Applied Ecology 41: 724-734.
- Appendix 3.10 Green, R.E., Langston, R.H.W., McCluskie, A., Sutherland, R. & Wilson, J.D. (2016) Lack of sound science in assessing wind farm impacts on seabirds. Journal of Applied Ecology, doi: 10.1111/1365-2664.12731
- Appendix 3.11 McGregor, R.M., King, S., Donovan, C.R., Caneco, B. & Webb, A. (2018) A stochastic collision risk model for seabirds in flight. Report to Marine Scotland.

Please could you kindly confirm receipt.

Best Regards

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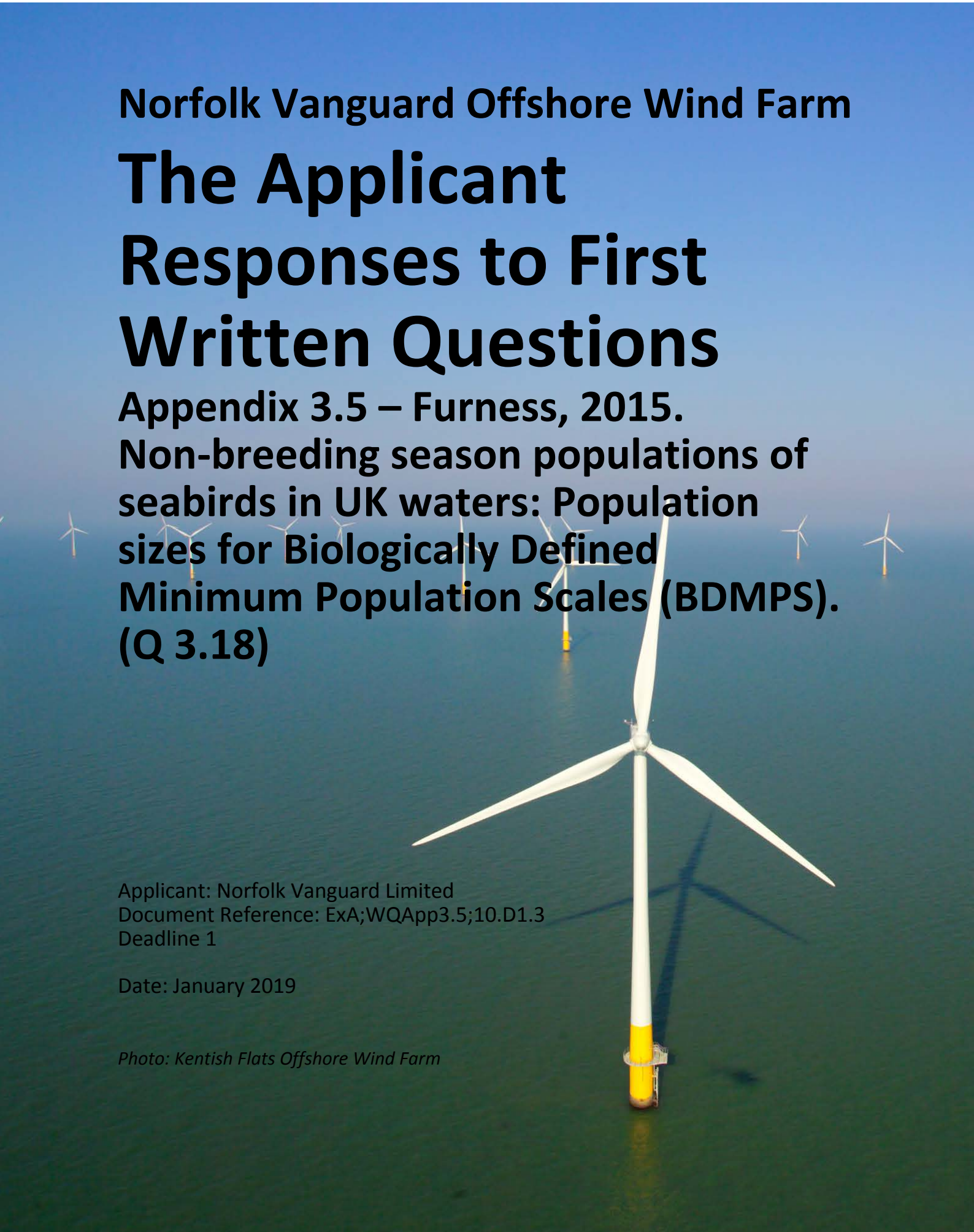
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VATTENFALL

Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.5 – Furness, 2015. Non-breeding season populations of seabirds in UK waters: Population sizes for Biologically Defined Minimum Population Scales (BDMPS). (Q 3.18)



Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.5;10.D1.3
Deadline 1

Date: January 2019

Photo: Kentish Flats Offshore Wind Farm

Natural England Commissioned Report NECR164

Non-breeding season populations of seabirds in UK waters

Population sizes for Biologically Defined Minimum Population Scales (BDMPS)

First published 22 January 2015

www.gov.uk/natural-england

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 current, unprecedented amount of marine renewables development proposed in UK waters have the potential to impact on seabird populations.

Impact assessments for marine renewables have tended to focus on potential impacts to seabirds during the breeding season when breeding birds are closely associated with their colonies, and where impacts can more easily be attributed to breeding populations, for example, based on foraging ranges. However, impacts to seabirds may also occur outside the breeding season, and to address the impacts of marine renewables across each species' full annual cycle, we need to determine a method for assessing seabird impacts during the non-breeding season. At the moment this is limited as we do not have agreed population scales or population estimates relevant to the non-breeding season to consider in assessments, and against which to apportion any non-breeding season impacts to populations or sites.

The aim of this project was to address this limitation by reviewing and defining species-specific non-breeding season seabird populations at biologically defined minimum population scales (BDMPS) to enable the apportioning of potential impacts of marine renewable developments during the non-breeding season.

The findings will be used by those engaged in marine spatial planning and impact assessments, in particular the Statutory Nature Conservation Bodies (SNCBs), regulators and developers in the offshore sector who need to assess the potential impact of offshore development proposals on seabird populations and protected sites for seabirds across their annual cycle.

The results will be used by the SNCBs to develop agreed methodologies on how to use the outputs of the work in the context of our advice to developers on assessing impacts to seabird populations from offshore developments.

This report should be cited as:

FURNESS, R.W. 2015. *Non-breeding season populations of seabirds in UK waters: Population sizes for Biologically Defined Minimum Population Scales (BDMPS)*. Natural England Commissioned Reports, Number 164.

MacArthur Green Ltd

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Keywords - environmental impact assessments (eia), habitats regulations assessment (hra), marine, non-breeding, offshore renewables, ornithology, populations, seabirds, special protection areas (spa)

Further information
This report can be downloaded from the Natural England website: **www.gov.uk/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

This report reviews evidence concerning the populations of seabirds that are present in UK waters during the non-breeding period. It uses the literature to assess the sizes of seabird populations with the aim to use the most up to date available data (usually expressed in terms of numbers of breeding pairs in each country). It uses data on the demography of seabirds (survival rates, age of first breeding, productivity) to model population age structure in order to assess the numbers of immature birds that are associated with breeding populations, since it is not normally possible to census immature components of seabird populations. Data on the timing of breeding and of migration are used to assess the appropriate seasonal definitions to use in this project; this assessment was based on literature and on appropriate data compliations such as annual bird reports, and online databases presenting seabird migration statistics. For each key species, migratory movements are reviewed based on literature and web pages reporting ring recovery data, geolocator tracking (for the few species for which tracking data are available), seawatching, at-sea survey data, biometrics and other markers of origins of birds. Numbers thought to be present in UK waters were also reviewed from these sources. Data on numbers of breeding pairs in UK Special Protection Area (SPA) breeding populations were tabulated for each species. Data were used to present hierarchical scales that can be of use in assessment of impacts on populations; firstly the biogeographic population with connectivity to UK waters (defining which populations visit UK waters and the estimated total numbers of birds (adults and immatures) in that combined population); secondly the total number of birds present in all UK territorial waters during the defined season; thirdly the total number of birds in each spatially distinct biologically defined minimum population scales (BDMPS) population during that defined season. BDMPS population sizes were estimated from the information reviewed on migrations of each population, and the most up to date data were used in an apportioning of birds from each population into each BDMPS. Confidence in the assessments of BDMPS population sizes was expressed using a traffic light coding where green represents numbers thought likely to be accurate to no more than 30% less or 50% more than the estimated number, amber represents numbers thought likely to be accurate to no more than 50% less or 80% more than the estimated number, and red represents numbers where the true value may lie more than 50% below, or 80% above, the estimate presented. It is intended that the apportioning tables (69 tables presented as Appendix A) can be updated as new census data become available, and as new data on migrations and winter distribution are gathered that allow more precise and accurate quantifications of proportions of populations present within defined spatial areas. A summary of the BDMPS populations is given in the following table. For details of defined spatial areas named in Table 0.1 see maps in each individual species' account.

Table 0.1. Summary of the estimated numbers of birds (adults plus immatures) in each BDMPS spatial and seasonal population for each seabird species considered in this report. BDMPS are colour coded to indicate level of uncertainty about numbers (green low, amber moderate, red high). See species accounts for details of uncertainty, including issues about numbers of BDMPS to be defined which are in addition to uncertainty about numbers.

Red-throated diver						
	Winter (December-January)		**Migration seasons (Sept-Nov and Feb-April)**			
	NW North Sea	1,523	UK North Sea	13,277		
	West of Scotland	861	UK western waters & Channel	4,373		
	SW North Sea	10,177				
	NW England & Wales	1,657				
	SW England & Channel	1,153				
Great northern diver						
	Non-breeding season (Sept-May)					
	West of Scotland	2,000				
	NW North Sea	1,000				
	SW North Sea & Channel	200				
	NW England & Wales	300				
	SW England	500				
Northern fulmar						
	Winter (November)		**Migration seasons (Sept-Oct and Dec-Mar)**			
	UK North Sea	568,736	UK North Sea	957,502		
	Western waters & Channel	556,367	Western waters & Channel	828,194		
Manx shearwater						
	Migration seasons (Aug-early Oct and late Mar-May)					
	UK North Sea	8,507				
	Western waters & Channel	1,580,895				
Northern gannet						
	Autumn (Sept-Nov)		**Spring (Dec-Mar)**			
	UK North Sea & Channel	456,298	UK North Sea & Channel	248,385		
	Western waters	545,954	Western waters	661,888		
Great cormorant						
	Non-breeding season (Sept-Mar)					
	NW North Sea	6,012				
	SW North Sea & Channel	10,460				
	West of Scotland	7,049				
	SW England & Wales	9,602				
European shag						
	Non-breeding season (Sept-Jan)					
	NW North Sea	45,503				
	SW North Sea & Channel	4,346				
	West of Scotland	37,363				
	SW England & Wales	13,075				
Arctic skua						
	Autumn (Aug-Oct)		**Spring (Apr-May)**			
	North Sea & Channel	6,427	North Sea & Channel	1,227		
	Western waters	5,287	Western waters	5,111		
Great skua						
	Autumn (Aug-Oct)		**Winter (Nov-Feb)**		**Spring (Mar-Apr)**	
	North Sea & Channel	19,556	North Sea & Channel	143	North Sea & Channel	8,485
	Western waters	16,336	Western waters	1,398	Western waters	25,090

Lesser black-backed gull						
	Autumn (Aug-Oct)		**Winter (Nov-Feb)**		**Spring (Mar-Apr)**	
	North Sea & Channel	209,007	North Sea & Channel	39,314	North Sea & Channel	197,483
	Western waters	163,304	Western waters	41,159	Western waters	163,304
Herring gull						
	Non-breeding season (Sept-Feb)					
	North Sea & Channel	466,511				
	Western waters	173,299				
Great black-backed gull						
	Non-breeding season (Sept-Mar)					
	UK North Sea	91,399				
	West of Scotland	34,380				
	SW and Channel	17,742				
Black-legged kittiwake						
	Autumn (Aug-Dec)		**Spring (Jan-Apr)**			
	UK North Sea	829,937	UK North Sea	627,816		
	Western waters & Channel	911,586	Western waters & Channel	691,526		
Sandwich tern						
	Migration seasons (July-Sept & Mar-May)					
	North Sea & Channel	38,051				
	Western waters	10,761				
Roseate tern						
	Migration seasons (Aug-Sept & late Apr-May)					
	East coast & Channel	251				
	N & W Scotland	4				
	W England & Wales	2,100				
Common tern						
	Migration seasons (late July-early Sept & Apr-May)					
	North Sea & Channel	144,911				
	Western waters	64,659				
Arctic tern						
	Migration seasons (July-early Sept & late Apr-May)					
	North Sea & Channel	163,930				
	Western waters	71,398				
Little tern						
	Migration seasons (late July-early Sept & mid-Apr-May)					
	North Sea & Channel	3,524				
	Western waters	1,602				
Common guillemot						
	Non-breeding season (Aug-Feb)					
	North Sea & Channel	1,617,306				
	Western waters	1,139,220				
Razorbill						
	Migration seasons (Aug-Oct & Jan-Mar)		**Winter (Nov-Dec)**			
	North Sea & Channel	591,874	North Sea & Channel	218,622		
	Western waters	606,914	Western waters	341,422		
Black guillemot						
	Non-breeding season (September-March)					
	N within 20 km					
Atlantic puffin						
	Non-breeding season (mid-August-March)					
	North Sea & Channel	231,957				
	Western waters	304,557				

1. INTRODUCTION

1.1 Background to this project

The UK Statutory Nature Conservation Bodies (SNCBs) – the Joint Nature Conservation Committee (JNCC), Natural England (NE), Natural Resources Wales (NRW), Scottish Natural Heritage (SNH), the Department of the Environment, Northern Ireland (DOENI) and Marine Scotland (MS) – require agreed population estimates for seabird populations in the non-breeding season. With recent Crown Estate leasing rounds, there is now an unprecedented amount of marine renewables development proposed in UK waters, all of which has the potential to impact on seabird populations, to a greater or lesser degree.

Current impact assessments for marine renewables focus on potential impacts to seabirds during the breeding season when breeding birds are closely associated with their colonies, and where impacts can more easily be attributed to breeding populations (e.g. based on foraging ranges). However, there is a need to consider potential impacts to seabirds outwith the breeding season, for which there is current lack of agreement on population scale and non-breeding season population estimates. These are required in order that non-breeding season impacts can be assessed, against appropriate populations.

To address the impacts of marine renewables across each species' full annual cycle, we need to determine the origins and sizes of seabird populations during the non-breeding season, and agree how to combine assessment of non-breeding season impacts with breeding season ones. As a first step, we require population estimates, at an agreed scale, for key seabird species (those most likely to be affected by development) occurring in UK waters in the non-breeding season. These then need to be adjusted to take account of immature birds present since those can form a high proportion of the population in species with deferred maturity. These regionally defined populations are the appropriate ones to consider for Environmental Impact Assessment (EIA). For Habitats Regulation Assessment (HRA), it is then necessary to consider which Special Protection Areas (SPAs) contribute to each regionally defined population.

1.2 Overall Aim

The overall aim of the project is to review and define species-specific non-breeding season seabird populations at biologically defined minimum population scales (BDMPS) to enable the apportioning of potential impacts of marine renewable developments during the non-breeding season. Species included in this review are: red-throated diver, great northern diver, northern fulmar, Manx shearwater, northern gannet, great cormorant, European shag, Arctic skua, great skua, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, roseate tern, common tern, Arctic tern, little tern, common guillemot, razorbill, black guillemot, and Atlantic puffin.

1.3 EIA and HRA non-breeding season assessments; project requirements

For EIA purposes, impacts need to be assessed against relevant regional populations, including not only birds from the UK but also birds from overseas populations that pass through UK waters on migration or winter in UK waters. This assessment can be at a range of spatial scales, from the biogeographic population downwards (biogeographic population scales have been well defined by JNCC and others – see for example Stroud et al. 2001; Kober et al. 2010, 2012; JNCC 2014). The largest spatial scale (the biogeographic population) is most easily defined in terms of seabird numbers and distribution, but would require cumulative assessment of all projects within the entire biogeographic population range which may be impractical. This report presents a smaller scale which is the

biogeographic population with connectivity to UK waters. That population is the sum of numbers in the UK population plus each overseas population known to visit UK waters either to winter or during migration to winter quarters elsewhere. That population is therefore in most cases smaller than the biogeographic population since the latter may include populations of the species that do not ever visit UK waters so are not at risk from development within UK waters. However, in many cases, overseas populations are large yet only a very small fraction of the population visits UK waters. So assessing impacts against the biogeographic population with connectivity to UK waters might assess the impact against much larger numbers than are ever present in UK waters. So the next step may be to consider assessment against the total number of individuals of the species that are present in UK waters at a particular season (non-breeding season, autumn migration, winter etc). For each species this total number, and the contribution of birds from UK and from overseas, is presented as a reference value. However, for Habitats Regulations Assessment (HRA) and to make EIA more practical, it may be more appropriate to consider smaller spatial scales, hence the derivation of the BDMPS. In general, for many species there are two clear BDMPS in UK waters, one in the UK North Sea and one in UK western waters. The UK land mass separates these areas and does present a significant barrier to movement of seabirds so that for most species the birds in the North Sea mix very little with birds in UK western waters and vice versa. For some species there are also clear separations between populations in northern and southern parts of UK waters. For a few species, while the two BDMPS seem appropriate, there is limited movement of birds from specific colonies within a BDMPS, so that a smaller spatial scale than the BDMPS would be appropriate for assessment. In such cases a more appropriate Reference Area needs to be defined in relation to a proposed development, taking account of the limited mobility of birds from focal colonies within the BDMPS.

The smallest spatial scale makes identifying all relevant projects much simpler, but comes at a cost of less clearly defined seabird populations as the exact movements in time and space of each age class of each population are not well known for any seabird species. For EIA, it is therefore likely that the optimal compromise is to define regional populations at an intermediate spatial scale between biogeographic and local. HRA requires that impacts to the proportion of the population that are qualifying features at SPAs are considered. This includes assessing the potential impact of offshore projects on SPA population features throughout the whole year. Where evidence allows, impacts to non-breeding season populations should be linked to specific breeding colonies. Where this is not possible, potential impacts might need to be assessed against the overall UK SPA network population of the respective species.

This requires the definition of the wintering area of UK breeding populations and an understanding of the influx of birds breeding abroad but mixing with UK SPA breeding birds within UK waters during the non-breeding season. As our understanding of biogeographic populations is relatively advanced, and breeding and non-breeding range for those biogeographic population units are more or less defined, the biogeographic population, which includes UK breeding birds, might represent the largest reference unit to start with in the absence of more specific knowledge.

Based on population estimates of the overall biogeographic population (e.g. AEWA (2012)) and the UK population of a specific species within the SPA suite, the proportional contribution of the UK SPA birds to a biogeographic population can be derived. Assuming an equal mixing of birds from across the biogeographic breeding range during the non-breeding season, this allows apportioning of potential impacts on the overall UK SPA network, or even to individual SPAs.

Nonetheless, whenever evidence allows, the aim should be to define non-breeding season biologically relevant population scales (BDMPS) which are smaller than the biogeographic

region. There could be a need to define different BDMPSs for different seasons. BDMPS may be appropriate for the entire non-breeding period for some species, or may need to be split into separate BDMPS for migration periods and for that part of winter when no migration occurs. The driver for all this is to generate a useful scale that allows us to determine what the impacts of an offshore project are on seabird populations and SPA populations. For this we need to be able to a) assign the birds that are predicted to be impacted by a project to a particular population and SPA; b) to calculate what proportion of the population and SPA population that number of birds represents.

With very few exceptions (such as penguins) seabird population sizes have never been counted. This is because, unlike penguins which moult communally with all age classes represented, most seabird species are never all in one place together. Data on seabird 'population' sizes are mainly presented in terms of numbers of breeding pairs, or in similar units (such as Apparently Occupied Territories) based on census work at colonies. However, these counts monitor only one part of the whole population (breeding adults). Seabird populations include not only these breeding pairs but also large numbers of sexually immature birds (because seabirds exhibit deferred maturity so immature birds can represent similar numbers to the breeding component), and in some cases some sexually mature non-breeding adults. Once the non-breeding season BDMPS is defined and the wintering population quantified, it is necessary to estimate the proportion of the population which do not contribute to the number of breeding pairs estimated at relevant breeding colonies. This will consist of large numbers of sexually immature sub-adults, and might in some cases also include sexually mature but non-breeding adults.

For the breeding season, the BDMPS is defined as the breeding population within foraging range from the project, plus non-breeders and immatures, which are likely to originate from a much wider range of colonies and may include young immature birds spending the summer in their wintering area as well as immatures loosely associated with local colonies. For the non-breeding season, the steps are as outlined above except for apportioning any impacts back to the SPA; separate BDMPS may need to be defined for the migration seasons as well as for the 'winter' period between migration seasons.

A literature review has been conducted to establish whether such proportions have been estimated for any of the priority species, and whether proportional estimates are appropriate to the BDMPS. Secondly, where such estimates have not been made, a review of demographic parameters has been undertaken to establish the most appropriate values to use for the BDMPS and indicate where data gaps exist, focussing on age at first breeding, productivity, and age/life stage-specific survival rates at suitable population scales. Thirdly, demographic parameters have been used to inform age-structured population models (e.g. Leslie matrices), to estimate a stable age distribution from which the proportion of breeding adults and of immature birds within the BDMPS can be estimated.

This report will soon become out of date. It will be necessary to update seabird population estimates and seabird movement patterns, to take account of new data and to take account of changes that are occurring as a consequence, for example, of changes in environmental conditions (such as distributions of fish stocks and fisheries management practices such as discarding). Furthermore, we will soon see new designations of Special Protection Areas for non-breeding seabirds. These new SPA designations are anticipated first to include inshore areas for non-breeding aggregations of divers, grebes and seaducks, and subsequently also marine areas for non-breeding offshore seabirds. Those designations have not been included in this report as the exact areas and species to be included remain uncertain at the present time.

2. METHODS

2.1 Breeding range and taxa

Information on the breeding range of the species, the number of sub-species that are recognised, and the breeding ranges of individual sub-species, was summarised from Forrester et al. (2007) and Brown and Grice (2005), with reference where necessary to Handbook of the Birds of the World (Hoyo et al. 1992-2011) and Birds of the Western Palearctic (Cramp et al. 1977-1994). Where sub-species are recognised, there is clearly scope to reduce the biogeographic population being considered to the relevant sub-species, and differences in the biometrics of different sub-species or populations within sub-species can also be informative about the origins of birds if their measurements can be obtained.

2.2 Non-breeding component of the population

Seabirds are generally long-lived animals which show deferred maturity. The species that are the focus of this project start to breed when, on average 2 (roseate tern) to 9 years old (northern fulmar). Therefore, a major part of the population will comprise immature birds. Seabirds are almost always censused in units of breeding pairs, so the population size based on breeding pairs provides only a partial census of the entire population. In this report the numbers of immature birds associated with breeding populations have been estimated by applying the simplest of Leslie matrix models to estimate the numbers of birds in each age class in a stable (equilibrium) model population (stable age distribution and immature survival rates adjusted to give a zero net rate of population change) with defined demographic parameters. Consistent as well as appropriate selection of demographic parameters is important. Therefore, for each species the age at first breeding and adult survival rate data presented by BTO Birdfacts (http://www.bto.org/about-birds/birdfacts accessed 11 March 2014) which presents the values considered to be most up to date and most appropriate for UK seabird populations were taken as the basis for modelling populations. This was done for consistency of approach and convenience, but with the caveat that BTO Birdfacts might not be the most appropriate source for all species and is only updated periodically. However, assessing all demographic data for all species would in itself represent a major project and was agreed to be outwith the scope of this project. Generally, most seabirds have been studied in enough detail to provide moderately precise measures of adult survival rate, although this can vary with colony size, food abundance and climate (Sandvik et al. 2012). Data on productivity (breeding success as chicks fledged per pair) were extracted as annual measures from each individual monitored colony from the JNCC seabird productivity monitoring database (http://jncc.defra.gov.uk/page-1550 accessed 11 March 2014) for the years 1986 to 2006 (the years for which data are presented in tables in annual reports). For the population model, data on age at first breeding, adult survival and mean productivity were used at face value. Data on juvenile or immature survival are not available for many seabird species, and those data reported in literature tend to be highly uncertain with very large confidence intervals and possible biases. So data on immature survival were used as a guide in constructing models, but survival rates input into the model were iteratively adjusted until the model produced approximate stability (a zero rate of population growth). This approach was considered to be precautionary in that an increasing population will tend to have a higher ratio of immatures to breeding adults than will be present in a stable population, whereas a declining population may or may not differ in ratio of immatures to adults depending on which age classes are exposed to elevated mortality rates that are causing the population decline. Adjustments of immature survival rates were made so that survival rates always increased with age up to the adult survival rate. Numbers in each age class were then used to estimate the ratio of immatures to breeding adults, making the (precautionary) assumption that no birds of breeding age took sabbatical years off breeding. In practice, it is known that in some seabird populations subject to extreme environmental stresses, some breeders will take sabbatical years, although for most species when conditions are normal or good, virtually all birds of

breeding age do breed (Harris and Wanless 1995). The age composition of populations is, of course, strongly affected by the mean age at first breeding. The BTO Birdfacts web site does not indicate the published sources from which they obtained data on mean age of first breeding for each species. However, their presented values appear to match with the literature, except in the case of Atlantic puffin. For puffins, Harris and Wanless (2011) indicate a mean age of first breeding of 7 years old, whereas BTO Birdfacts cites a mean age of first breeding at 5 years old. Modelling the population using an age of first breeding of 5 years generates an estimated 0.82 immatures per breeding adult whereas for an age of first breeding of 7 years generates an estimated 1.08 immatures per breeding adult (making no changes to the productivity and adult survival rates used in the model). This is likely to be the largest uncertainty in the estimated ratio of immatures to adults, as the age of first breeding seems to be better known for most other seabirds.

Implications of altering adult survival rate for the ratio of immatures per adult are generally moderate (Figure 2.1), as are implications of altering age at first breeding (Figure 2.2) or productivity (Figure 2.3). The proportion of immatures tends to decrease with increasing adult survival rate, but tends to increase where age at first breeding increases, and tends to increase with productivity of the population.

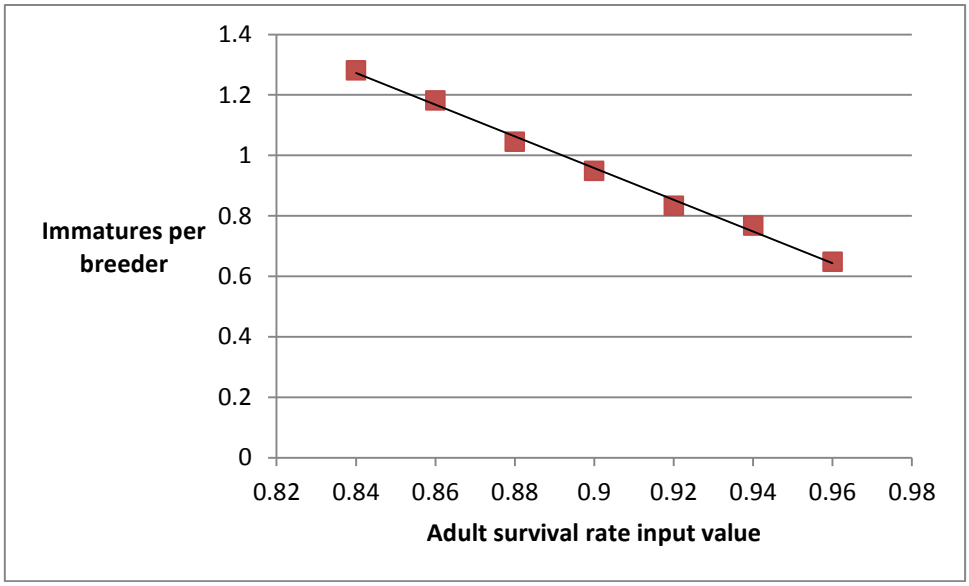


Figure 2.1. Model estimates of the numbers of immatures per breeder (ranging from 0.65 to 1.28) for a range of values of adult survival rate (from 0.84 to 0.96), values of productivity and age of first breeding being held constant at mean values.

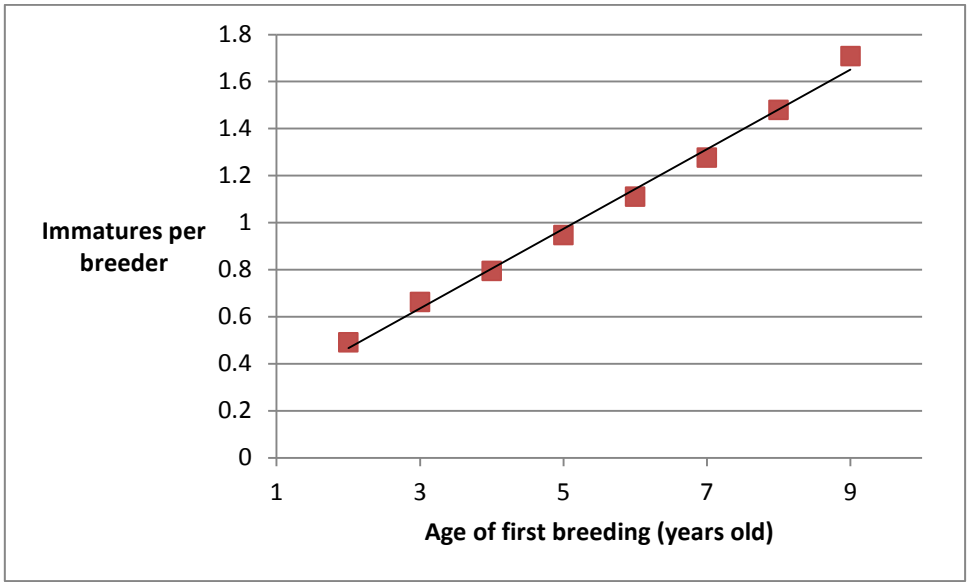


Figure 2.2. Model estimates of the numbers of immatures per breeder (ranging from 0.49 to 1.71) for a range of values of age of first breeding (from 2 to 9 years old), values of productivity and adult survival being held constant at mean values.

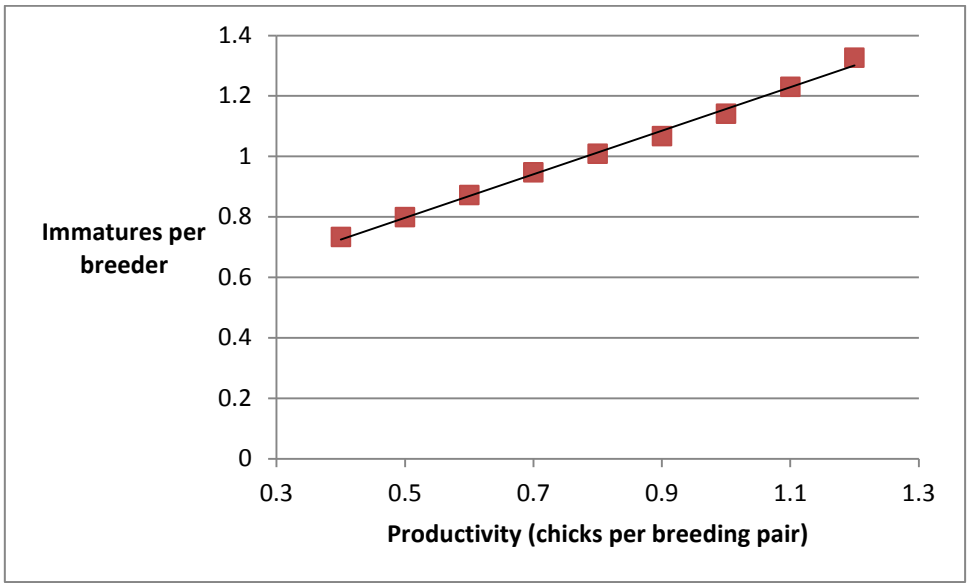


Figure 2.3. Model estimates of the numbers of immatures per breeder (ranging from 0.73 to 1.32) for a range of values of productivity (from 0.4 to 1.2 chicks per pair), values of age of first breeding and adult survival being held constant at mean values. Note that in all of these analyses, the ratio of immatures to breeders is close to 1, meaning that under a range of plausible demographic values seabird populations contain a similar total number of immature birds to the total number of breeding adults.

Age of first breeding, adult survival rate and productivity data input into the Leslie Matrix model are summarised below, together with the derived estimate of the number of immatures per breeding adult in a typical population of each species.

Species	**Age of first breeding (years)**	**Adult survival rate**	**Mean productivity**	**Estimated immatures per breeding adult in population**
Red-throated diver	3	0.84	0.635	0.74
Great northern diver	6	0.9	0.635	1.1
Northern fulmar	9	0.972	0.424	0.62
Manx shearwater	5	0.905	0.591	0.84
Northern gannet	5	0.92	0.684	0.81
Great cormorant	3	0.88	1.913	1.17
European shag	4	0.878	1.289	1.31
Arctic skua	4	0.886	0.522	0.71
Great skua	7	0.888	0.664	1.42
Lesser black-backed gull	4	0.913	0.517	0.68
Herring gull	4	0.88	0.936	1.09
Great black-backed gull	4	0.88	1.139	1.26
Black-legged kittiwake	4	0.882	0.672	0.88
Sandwich tern	3	0.898	0.656	0.63
Roseate tern	2	0.855	1.293	0.75
Common tern	3	0.9	0.721	0.67
Arctic tern	4	0.9	0.402	0.58
Little tern	3	0.899	0.521	0.56
Common guillemot	5	0.946	0.678	0.74
Razorbill	4	0.9	0.633	0.75
Black guillemot	4	0.87	1.295	1.32
Atlantic puffin	7	0.93	0.67	1.04

Although modelling shows that the ratio of immatures per adult is relatively robust to errors in demographic parameter estimates, it would be useful to be able to validate these ratios. Data on the proportions of different age classes of seabirds at sea might seem to be one way to try to do this. However, very few seabird species can be identified to age classes with confidence. For example, ageing of auks at sea is almost impossible. Kittiwakes can be identified as juvenile/first year or 'adult' based on plumage, but the 'adult' category will include many immatures as well as birds of breeding age. Large gulls can be more securely aged based on plumage, but there is considerable overlap in plumages between age classes and older immatures are not easy to separate from adults in the field so that survey fieldwork that is not specifically aimed at determining numbers of each age class is likely to mis-classify many individuals. Gannets have a sequence of plumages that allow fairly detailed classification of birds into ages, but again the older immatures can be mistaken for adults if not examined in detail. Moreover, the at sea distribution of seabirds differs between age classes, with youngest birds tending to spend their time in the winter quarters even during summer, breeding adults tending to stay closest to their breeding area, and immature birds probably at sea in areas that have good food supplies but are away from large colonies. So it is not clear that any at sea data on proportions of different age classes would provide a secure test of the estimated proportions based on demographic data.

2.3 Phenology

Information on the timing of seabird breeding seasons (initial arrival back at the colony in spring, modal return to colony in spring, modal departure from colony at the end of the breeding season, and final departure from the colony) was extracted from Forrester et al. (2007) and Pennington et al. (2004). In addition, data on modal arrival at colonies in spring, and modal departure from colonies in autumn were extracted from Orkney Bird Reports for 2008, 2009, 2011 and 2012, Shetland Bird Reports for 2008 to 2012, and Fair Isle Bird Observatory Reports for 2007 to 2012. These data were collated in an Excel spreadsheet and then used to describe the timing of seabird breeding seasons. Data on timing of seabird migrations were obtained from several sources. Timings for autumn and spring migrations (beginning/peak/end), were extracted from Cramp et al. (1977-1994), Wernham et al. (2002), Pennington et al. (2004), Brown and Grice (2005), Forrester et al. (2007) and Vanermen et al. (2013). The earliest spring sighting of the species, peak of spring migration, peak of autumn migration, and the last reported sighting of the autumn were extracted from Orkney Bird Reports for 2008, 2009, 2011 and 2012, Shetland Bird Reports for 2008 to 2012, Fair Isle Bird Observatory Reports for 2007 to 2012, Argyll Bird Reports for 2008 to 2012, and the Gibraltar migration watch website www.gonhs.org. In addition, data for each seabird species on the mean numbers per hour observed at UK migration sites for each week of the year (averaged over all years for which data were collected) were extracted from the Trektellen migration web site www.trektellen.nl. The Trektellen data were used to plot histograms describing the seasonality of observations at migration sites (most of which are located in E or SE England), to infer the timings of spring and autumn migrations. As with timing of breeding, extracted data on timing of migrations were entered into an Excel spreadsheet and these data were then used to describe phenology in individual species accounts. The timing of breeding seasons defined within this report is evidence-based from the literature, but does not replace SNCB guidance documents on seabird breeding seasons. Where SNCB guidance differs from the seasons presented in this report, the definitions of seabird breeding and non-breeding periods in assessments needs to be agreed with SNCBs in advance of assessments being made.

Accounts of phenology in Cramp et al. (1977-1994) differ somewhat from those in Wernham et al. (2002), Pennington et al. (2004), and Forrester et al. (2007), in that Cramp et al. consider the timings of migrations throughout the species' range and not specifically in UK waters. For that reason, less attention was given to details in Cramp et al. (1977-1994) except where this either did, or did not, match up with data in the other sources. Data on

phenology in Wernham et al. (2002) were sometimes equivocal, as it was not always evident whether text was describing seabird phenology or the phenology of ring recoveries from seabirds; phenology indicated by the timing of ring recoveries is likely to be biased by the fact that recoveries may occur some days, or even weeks, after the death of the bird rather than at the exact time of death. For this reason, more weight was given to the phenology data in Pennington et al. (2004), Brown and Grice (2005), and Forrester et al. (2007). Data in Bird Reports provide accurate and detailed information on the first arrival dates of spring migrants, and fairly detailed and accurate data on the last sightings of the year of departing autumn migrants, but provide less information on modal migration dates. Data from Pennington et al. (2004) were compared with data from Forrester et al. (2007) and Brown and Grice (2005) to see whether there was a detectable progressive difference in timing between the north and south of the UK. Data from Gibraltar and Belgian migration studies reported in www.gonhs.org and Vanermen et al. (2013) were used as context, specifically to test whether there were clear differences in phenology between the UK, and regions south of the UK. For almost all species, differences in timing between years (Frederiksen et al. 2004, 2013), and differences reported by different authorities were as great as, or greater than, any slight differences in timing between latitudes within the UK, so to avoid excessive complexity, summaries of phenology were derived for all UK waters rather than for separate regions.

2.4 Defined seasons

Seasons were defined for each species as 'breeding season' and 'non-breeding season' in the context of UK breeding. Breeding season was defined as the period from modal return to the colony through to modal departure from the colony at the end of breeding, for birds at UK colonies. Breeding season was defined as the period between modal return of breeding adults to colonies in 'spring' to modal departure from colonies at the end of the breeding season. Modal date is roughly equivalent to mean or median date, but is used here for pragmatic reasons – mean or median dates are difficult to measure and are rarely reported in the literature, whereas modal date is frequently reported. Use of first or last dates was avoided since extreme cases can be very misleading and atypical, and tend to vary with sample size. Non-breeding season was defined as the remaining part of the year.

Post-breeding (autumn) dispersal/migration, and pre-breeding (spring) migration periods were also defined, based on the periods during which substantial migration of the species occurs through UK waters. Therefore, the migration periods may overlap with the UK breeding season and with the non-breeding season, since timing of migrations of birds from high latitude regions can differ from that of UK birds. Wherever possible, seasons were defined as a set of months rather than in any more precise terms. This reflects the fact that for many seabird species phenology can vary by several weeks from year to year, so that greater precision is inappropriate. It also acknowledges the fact that survey work is normally carried out by calendar month, so that splitting survey data by periods shorter than one month can be inconvenient and technically difficult. However, for some species, especially long distance migrants such as terns, phenology is highly predictable and occurs within a narrow window. In such cases subdivision into fractions of months is appropriate, and has been done where necessary.

Spring migration for each species was defined as the months during which migratory movements of the species through UK waters towards breeding colonies (whether UK colonies or colonies of overseas populations) was clearly evident. Thus, spring migration may overlap with either or both of the non-breeding season and breeding season. Autumn dispersal/migration for each species was defined as the months during which migratory movements of the species through UK waters away from breeding colonies (whether UK colonies or colonies of overseas populations) was clearly evident. Thus, autumn dispersal/migration may overlap with either or both of the non-breeding season and breeding

season. Rationale for choice of months to define in each season is presented in the text for each individual species account, and each species' account is based on the extracted bird report phenology data tabulated in Excel, in addition to the broad literature review.

2.5 Movements of birds from the UK population through UK waters and from overseas populations into or through UK waters

A number of approaches can provide data on seabird migrations, including seawatching from coastal sites, recoveries of ringed birds, deployment of geolocators or other tracking devices, interpretation of biometrics, genetics, stable isotopes and other markers. The utility of these methods is outlined below.

2.5.1 Seawatching

Data from seawatching sites provides information on phenology of movements, but relatively little information on where birds come from or are going to. However, combined with other methods, seawatching data can provide supporting evidence of the seasonal movements of seabirds. The Trektellen web site www.trektellen.nl provides data on rates of movement of seabirds past UK (and other European) migration sites. Seawatching data are also used by Forrester et al. (2007) to assess numbers of seabirds migrating through Scottish waters. Seawatching data do not necessarily provide a good measure of numbers of birds as the counts one day may, or may not, involve the same individuals seen on a previous day. This can give a misleading impression. In general, numbers recorded on spring migration tend to be smaller than on autumn migration. While there will be smaller numbers migrating through UK waters in spring (in part because there will be many juveniles in the autumn passage but few in the spring return passage because most remain in winter quarters for their first summer and some do not survive the winter), another likely explanation of this is that migration in autumn can be a slow process with birds stopping off to feed at suitable sites on their way through UK waters, whereas in spring the adults migrate rapidly back to their breeding site because there is potentially competition for nest sites and a bird arriving back late may miss out. As a result, counts in autumn on any one day may be larger than in spring because birds remain on autumn passage for days or weeks, compared to the rapid flight through in spring. Tracking studies provide some support for this impression of more leisurely migration progress in autumn than in spring, but do not yet provide an accurate quantification of this difference.

2.5.2 Ringing data

Much of our understanding of seabird migrations is based on recoveries of ringed (and in some cases colour marked) seabirds. Ring recovery data were summarised for each bird species occurring in the UK by Wernham et al. (2002). A migration atlas has also been published for the Faroes (Hammer et al. 2013). Numerous papers have been published describing details of the seasonal movements of particular species of seabirds. Ring recovery data have many potential biases. Ringed birds are very unlikely to be recovered in the open ocean. Dead seabirds can be carried large distances by currents and can be deposited onto beaches far from where they died. Reporting probability can be high in countries (such as Greenland) where many seabirds are hunted for food. Recoveries of seabirds may be associated with fisheries bycatch or oil pollution incidents. Large numbers of seabirds may be ringed at a few colonies but none at other colonies (for example, most gannet ringing has been done on the Bass Rock, and for obvious reasons none or very few have been ringed at most of the gannet colonies where safe access to nests is impractical). Large numbers may be ringed in some countries but not in others. Interpretation of the migration routes and wintering areas of seabirds has to be done with great caution, trying to take account of these potential biases in data. Fortunately, these biases are well recognised and can mostly be taken into account, although the magnitude of the bias may not be easy to assess in some cases, especially in relation to the more pelagic seabird species. Ring

recovery data have been used as key information in this project to assess movement patterns of UK seabirds and to assess origins of seabirds that winter in, or migrate through UK waters. The Migration Atlas (Wernham et al. 2002) has been used as the most important source of information on this topic for most species of seabird, supplemented by more recent publications on seabird migration (which are predominantly single-species studies). Other especially useful accounts providing coverage of most species of seabirds include the Faroese Migration Atlas (Hammer et al. 2013), and species accounts in the book on seabirds in the Barents Sea (Anker-Nilssen et al. 2000).

2.5.3 Geolocation data loggers and other tracking devices

In recent years, new technologies have become available that can be used to study seabird migrations and wintering areas. In particular, several studies have deployed geolocation data loggers on breeding seabirds at various colonies. Geolocation data loggers are very small devices that can be attached to a leg ring on a breeding seabird of moderate size. About a year later, if the bird can be recaptured, the logger can be removed and data downloaded for analysis. These loggers record light intensity, and usually also temperature, on a time base. In principle, analysis of location from these data is simple. In the northern hemisphere, daylength is longer further north in summer, shorter further north in winter. At any given latitude, sunrise occurs earlier further east. Using light intensity data recorded in the logger, the location of a bird can be estimated twice each day from the light data (Phillips et al. 2004). In some situations, temperature data can help with estimating location (Teo et al. 2004) since the temperature recorded will be sea surface temperature when the bird is sitting on the water (which many seabirds always do at night when away from the colony). Location estimates are imprecise. The average error is around 180 km (Phillips et al. 2004, Teo et al. 2004). But this is adequate to establish the general area in which the bird is present. Geolocation does not work at the equinoxes, but this results in the loss of only a few weeks of data at those times of year (although those periods may well be during active migration by many species). Logger data can also be used to infer behaviour of birds, especially amounts of time spent flying, and spent sitting on the water (Mackley et al. 2010).

2.5.4 Biometrics

Many seabirds show variation in biometrics between populations. In many cases birds breeding further north tend to be larger in size. Biometrics can be used to infer origins of those seabirds that show clear and known variation in measurements between populations. This has been used very successfully for great northern divers, and to some extent for auks. There are probably several species of seabird where biometrics could be informative but there has not yet been an assessment of the use of this approach. There are, however, some seabirds where biometric variation between populations appears to be too small to be useful. There are also difficulties created by post-mortem shrinkage (e.g. Harris 1980), and variability in measurements recorded by different researchers, some, but not all, of which are due to differences in measurement technique (Barrett et al. 1989).

2.5.5 Genetics

There are a few phenotypic features of seabirds that show clinal variation with latitude, and so have potential to provide information on the breeding season origins of birds sampled in winter. While most Arctic skuas at lowest latitude breeding areas are dark phase birds, the proportion of light phase increases northwards and reaches 100% on Arctic tundra. The proportion of dark phase fulmars increases with latitude in the North Atlantic. The proportion of ‘bridled’ common guillemots increases with latitude. Herring gulls from high latitude colonies tend to have more white on the tips of the outer primaries than seen on birds from low latitude colonies, and also have darker grey mantle plumage. A number of studies have investigated whether molecular genetic markers, such as mtDNA, can be used to identify breeding colony or regional origins of seabirds sampled outside the breeding season, but these studies have not generally been very successful in identifying specific genetic markers

that can be used in this way. However, it seems likely that some suitable genetic markers might be identified in future.

2.5.6 Stable isotopes and other natural markers and pollutant markers

Stable isotopes of carbon and nitrogen can be measured in feather samples. These ratios tend to reflect diet at the time of feather growth, which for juvenile seabirds is at the breeding site, whereas for most species of seabirds the moult of adults occurs in the wintering area, though there are certain exceptions (Cherel et al. 2006). Leat et al. (2013) recently showed that the carbon isotope ratio in feathers of great skuas sampled at breeding colonies is indicative of whether individual breeding adults overwintered off west Africa, or off southern Europe, or off North America. That study also identified characteristic differences in the proportions of different persistent organic pollutants in birds, reflecting which of these three regions the individual used as its wintering area. Similar differences have been seen in feathers of gannets and lesser black-backed gulls that could be used to identify which individual birds had spent the winter off west Africa and which had wintered in European waters (the difference in carbon isotope being determined by the upwelling oceanography off west Africa which creates a distinct carbon isotopic signature in the food web that is clearly different from that found in European shelf seas). The use of isotopes, pollutants and other markers (such as heavy metals in feathers) as tracers of the origins of individual seabirds almost certainly has the potential to be developed in future, but has not yet been investigated in enough detail to be used to assess existing data except in a very few cases.

2.6 Numbers in UK waters

At sea surveys include the European Seabirds at Sea (ESAS) database which holds information on numbers of seabirds at sea counted using standardized methodology (Tasker et al. 1987). These data can be used to estimate seabird densities at sea in different months and locations and hence can provide evidence of seasonal changes in distribution (Tasker et al. 1985). ESAS data primarily relate to the North Sea and data are predominantly from the 1980s with fewer surveys in recent years, but ESAS methods have also been used in marine areas west and south of the UK and to some extent in years from the 1990s to the present. The ESAS data have been used to identify areas that may qualify as Special Protection Areas for seabirds on the basis of high densities of key species at particular times of year (Kober et al. 2010, 2012). However, Kober et al. (2010) were cautious about interpreting the absolute magnitude of density estimates from the ESAS data and chose to make corrections to absolute numbers of some species in order to make them match to ICES published data on numbers of seabirds in European waters. Some of the 'rescaling' factors quoted by Kober et al. (2010) were large. The accuracy of these 'rescaling' factors is rather uncertain, but suggests that the ESAS data provide only indications of relative abundance in different areas rather than meaningful measures of absolute abundance of seabirds at sea. WWT Consulting (2013) combined the ESAS data together with WWT aerial survey data to describe seabird distributions within English territorial waters. That exercise makes use of more recent survey data and allows aerial survey data to be included as well as boat-based survey data. That work also indicated significant discrepancies between data sets from aerial and from boat-based surveys (WWT Consulting 2013). However, the data were adequate to map seabird relative density across large areas of UK waters, and the methodology has recently been presented in Bradbury et al. (2014), and this represents the best available dataset for assessment of seabird distribution and relative abundance in UK waters during the non-breeding season. However, recognising the uncertainty about absolute numbers estimated from ESAS data and the somewhat out of date nature of that database, in this report, estimates of seabird density and distribution from ESAS and publications based on that database have been used primarily to provide a sense check on numbers considered to be in UK waters based on knowledge of population sizes and migration behaviour, rather than as a tool to define BDMPS totals.

2.7 Biogeographic populations

Starting from the Biogeographic population defined by Stroud et al. (2001) each species specific appropriate Biogeographic population was refined by considering only those populations with connectivity to UK waters at some time of year based on ringing and tracking data and the most recently available data on population sizes in the relevant countries (the latter primarily from Mitchell et al. 2004 but taking account of more recent publications where available – see individual species accounts for details). Estimates of breeding numbers in the UK were taken from Mitchell et al. (2004) as the most recent comprehensive surveys of most species, updated if possible by more recent survey data (such as national gannet surveys, skua surveys in Orkney), and data presented by SNH (Foster and Marrs 2012) or JNCC online seabird database. Amongst other sources, numbers were taken from the review by Lewis et al. (2012) but these data need to be treated with caution as SNH have found that numbers in that report are sometimes based on incorrect boundaries and population estimates. Numbers can be expressed in terms of the normal census unit (breeding pairs or equivalent such as (Apparently Occupied Territories (AOTs) or Apparently Occupied Nests (AONs)), or as the total population including numbers of immatures associated with a breeding population of the estimated size (based on the ratio of immatures to breeding adults estimated from the simple population model).

For those species where possible, data from the JNCC seabird population monitoring database were used to graph the breeding population trend from 1986 to 2012 in monitored UK colonies. As default, a linear trend line was fitted to these data, but where a non-linear trend provided a significantly better fit to the empirical data, a non-linear trend is presented, with the equation of the trend line and the amount of variance explained by the trend also presented on the graph. These trend lines have not been used to adjust count data for individual populations to bring it up to date, although such extrapolations would be possible if felt desirable in specific cases. The objective of presenting trends (which are shown for regions of the UK when the data allow and trends show different patterns in different regions) is to provide context that may be useful in the interpretation of BDMPS data and the understanding of how UK seabird populations may be changing in breeding numbers.

2.8 Proportion of UK population from UK breeding SPAs

The proportion of the UK population of each species that represents birds from UK SPAs with that species as a feature (including all those listed in JNCC 2014) was estimated by reviewing literature to obtain the most up to date available count of breeding numbers of each species at each SPA. JNCC (2014) provides an estimate of the proportion of the breeding population that is in SPA breeding sites, focused on the time period around 2000-2005 (since many colonies, especially non-SPA colonies, have not been counted since Seabird2000). However, for many SPAs, data are available for years since 2005. In many cases, the most up to date data were found on the JNCC Seabird Colony Monitoring web site database http://jncc.defra.gov.uk/smp/. Where there was evidently more recent data available for a site but those data were not entered into the SCM database, requests were made to access those data. For example, red-throated diver breeding numbers are not included in the SCM database but the SNH Sitelink web page indicated that Site Condition Monitoring data existed for some sites that were not available in published literature, and these were obtained from SNH staff. David Stroud at JNCC kindly provided access to the forthcoming JNCC SPA 2014 review to check that most recent survey data presented in this report match those used in the JNCC SPA 2014 review (JNCC 2014). That review also provides an estimate of the proportion of birds breeding in UK SPAs during the period around 2005. Where there have been no recent surveys of seabird numbers at particular SPAs, national, or where available regional, breeding population trends were obtained from the JNCC Seabird Numbers and Productivity database http://jncc.defra.gov.uk/page-1550. These trend data could be used to extrapolate numbers from the historical data to the present based on the estimated population trend at regularly monitored colonies; this

approach could be used for individual SPA sites where recent count data are not available, and could be used for national/regional numbers. There are potential difficulties using data from a subset of sites to extrapolate either for individual SPA sites or regional population size, since the set of monitored sites may not be representative of an individual site or of the entire population. This approach may only be required for a small number of cases, as many populations have been surveyed regularly so up to date data are available. For example, almost all tern colonies are monitored annually. In a very few cases (for example for great skua), trend data were not available from the JNCC monitoring programme. In such cases a trend could be estimated from available data from other SPA populations of that species. Where relevant, such issues are detailed in individual species accounts. However, to provide transparency in this report, the most recent counts are used for each population (and are listed in detail) so that subsequent work could make use of these counts (updating them or applying trend data to refine estimates as felt appropriate). However, in this report the use of trend data to correct population estimates has generally been avoided because it is often uncertain which trend would be appropriate to use, and there is evidence that trends at individual colonies often do not follow national or regional trends. So applying corrections to update old survey data is tempting, but the temptation has been resisted in order to present best available data rather than adjusted data.

Therefore, numbers presented in Appendix A Tables 1 to 69 are the most recent available counts for each colony or national population. The key exception to this rule is the estimate of numbers of pairs at non-SPA colonies in the UK where census data are generally not available since Seabird2000. In that case, for a few species where large changes in numbers are known to have occurred, the total in non-SPA colonies has been estimated to a value that approximately retains the proportion breeding in SPA populations at the value defined by the JNCC 2014 SPA review (JNCC 2014), and is consistent with the national or regional trend in breeding numbers reported by Foster and Marrs (2012) and the JNCC Seabird Monitoring Programme. Where this correction has been applied it is clearly indicated as a footnote to the tables in Appendix A.

Adjustment of old SPA count data allowing for trends would alter the estimated BDMPS slightly, but in practice there are few seabird SPA populations in the UK that have not been counted since Seabird2000, and trend adjustment would make only rather small differences to BDMPS totals relative to the influence of other factors such as estimation of the proportions of overseas populations entering UK waters or the sizes of overseas populations. There may be a case for employing trend adjustments of old count data where HRA is assessing impacts on specific SPA populations where data are old, but for EIA and for HRA where the colony is not the focal colony in an assessment, correction of old data is probably undesirable in most cases.

2.9 Appropriate BDMPS populations

Where the proportion of each population that occurs in UK waters is known, the Biogeographic population estimate can be narrowed to the numbers occurring within defined UK waters, creating Biologically Defined Minimum Population Sizes (BDMPS). The BDMPS spatial area is from the UK coast to the edge of UK territorial waters, bounded by defined lines running from selected points on the coast to the UK waters limit. The justification for having more than one BDMPS in UK waters is that there may be good evidence that the overall number of birds or the population origins of a particular species differ between areas. In that case estimating the impact that might be attributed to a particular SPA population whose birds occur within a development area depends upon identifying and using in apportionment the estimate of the appropriate number of birds which may be represented at a particular time of year in that sea area. Using different figures in different parts of UK waters is justified only if the overall suite of birds passing through the area is known to be different to that in another area. Thus, for example, red-throated divers in the southwestern

North Sea originate predominantly from continental European populations with only a small minority of birds from UK populations, whereas red-throated divers in the northwestern North Sea originate predominantly from UK populations, with only a minority of birds coming from overseas populations. These areas are therefore more appropriately considered as separate BDMPS.

Where the BDMPS is over an inconveniently large area, and especially where it is known that birds from specific colonies tend to remain within only a part of the BDMPS, it may be possible to define geographical reference regions that are convenient but not necessarily distinct in terms of the biogeographic populations present. That approach may be necessary for a few species, where populations are not very mobile but are distributed in overlapping areas across a much larger spatial scale. An example of this is common guillemot, where there are numerous SPA populations distributed from northern Shetland to the southern North Sea in a continuum, yet birds from particular SPA colonies are not distributed over the whole region but tend to remain nearer to their colony. It is therefore difficult to establish BDMPS boundaries within the whole region, but an assessment of impact needs to consider the localised movements of birds from particular colonies. In these cases, it may be necessary for HRA assessment to consider only the part of the BDMPS which would have connectivity with any particular development site rather than the entire BDMPS (so defining a specific 'reference region' that would be project-specific). The proportions of birds of a particular species present in each BDMPS or reference region can be estimated from information in the literature on seabird numbers and distribution, and from the evidence on the migrations of birds from defined populations. The allocation of numbers of seabirds from overseas populations migrating through, or wintering in different regions is rather uncertain for most seabird species, and in almost all species is much less well known than for UK populations. In a few cases, there are clear distribution patterns and well defined numbers of birds, but in most cases the numbers in different regions are not well defined, and movement patterns of immature birds are not known except in a very general way. This represents a major constraint on assessing the proportions of birds in UK waters from different overseas populations. Although numbers are often uncertain, calculations need to be made using best available data and explicit assumptions. Therefore the computations involved in establishing BDMPS totals are presented in Appendix A Tables 1 to 69. It is assumed that these working tables can be updated as new information becomes available to make estimates of BDMPS and the contributions of individual SPA populations to these BDMPS more up to date and more accurate. It has to be recognised however, that while numbers can be added together to achieve a total for the BDMPS, there is much uncertainty about the values being summed, and that the resulting BDMPS has a large, but also uncertain, confidence interval. For this reason, BDMPS estimates in the report are colour coded green, amber or red, according to the uncertainty, with a narrative explanation of the colour coding given below the summary table at the start of each species account. For estimates that are coded green, the numbers are likely to be no more than 30% less or 50% more than the estimate presented. For estimates that are coded amber the numbers are likely to be no more than 50% less or 80% more than the estimate presented. For estimates coded red the numbers might be more than 50% less or 80% more than the estimate presented. While these ranges are expert judgement based on the literature reviewed in this project, it is impossible to measure the uncertainty and so no confidence limits can be quantified. Therefore, the colour coding itself can only be considered indicative based on available knowledge, and should not be used to estimate confidence limits for BDMPS population estimates.

For some seabirds, such as Arctic skuas, terns and Manx shearwaters, there is no need to derive winter BDMPS on the grounds that to all intents and purposes these species are absent from UK waters at that time of year.

2.10 Proportions of birds from UK SPA populations in each BDMPS

Using the best available data of numbers of birds in UK SPA populations and taking account of associated numbers of immature birds, numbers in non-SPA colonies and numbers from overseas populations, once the size of a BDMPS population has been estimated, it is possible to estimate the proportion of those birds in the BDMPS originating from each individual UK SPA population, as required for HRA. This estimate will be very imprecise where details of population sizes or migratory movements are not well known, which unfortunately is the case for many seabird species.

2.11 Spatial distribution of UK breeding SPA birds across the BDMPS

For most seabirds, SPAs have been selected to give a good geographical representation of the species' protected breeding sites within the UK, so that the distribution of SPAs reflects the distribution of the population as a whole. This is particularly the case where the SPA populations sum to a high proportion of the total population. For relatively few seabird species, the distribution of SPA populations may not closely reflect the overall distribution pattern. Where this might be the case the distribution of SPA populations is assessed in relation to the overall distribution of the breeding population.

2.12 Presentation of BDMPS data in this report

Each of the species accounts that follows in this report starts with presentation of summary data outlining:

a) The biogeographic population with connectivity to UK waters (total number of birds including adults and immatures) and the contribution from UK and from overseas populations;
b) The number of birds (adults and immatures) in the whole of UK territorial waters within each distinct seasonal period relevant for that species and the contribution from UK and from overseas populations;
c) The number of birds (adults and immatures) in each separate BDMPS defined for that species in each distinct seasonal period relevant for that species and the contribution from UK and from overseas populations.

This hierarchical approach provides the opportunity to consider the relevant population scale for EIA, from biogeographic to BDMPS. It seems likely that the BDMPS population would be the most appropriate scale for use in EIA assessment in most cases, though the greater confidence in numbers at higher levels in the hierarchy could provide grounds for considering use of a higher level population scale in some cases.

Each of these totals is colour coded using the traffic light system, with reasons for the colour coding outlined in text below the summary table. For estimates that are coded green, the numbers are likely to be no more than 30% less or 50% more than the estimate presented. For estimates that are coded amber the numbers are likely to be no more than 50% less or 80% more than the estimate presented. For estimates coded red the numbers might be more than 50% less or 80% more than the estimate presented. The data on which these totals are based is presented in detailed tables (Appendix A Tables 1 to 69) which give the most recent count of each SPA population size, non-SPA population or overseas population (breeding pairs) on which the BDMPS numbers are based, the computed total number of adults, the corresponding total number of immatures, and the proportion of each population estimated to be present in each BDMPS and the resulting total number of individuals (adults, immatures and all ages). These data tables are likely to be used in assessments apportioning impacts of developments on particular populations for EIA and especially for HRA assessments. The data could be updated in each table as new data become available, and updates could include not only updating of population counts but also updating of proportions present in the BDMPS as new information on migrations becomes available.

Numbers in the BDMPS population estimate are given to the nearest individual bird because those totals are the sum of calculations presented in Appendix A Tables 1 to 69. However, the presentation of those totals to the nearest bird does not indicate high accuracy and comments on uncertainty in the BDMPS estimates should be considered with care.

3. RED-THROATED DIVER *Gavia stellata*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in winter (December-January) (adults and immatures)	Numbers in UK waters in migration seasons (September-November and February-April) (adults and immatures)
Overseas	**22,600**	**12,079**	**13,375**
UK	**4,400**	**3,292**	**4,275**
Total	**27,000**	**15,371**	**17,650**

Winter BDMPS (December-January)	**Total number of birds in BDMPS (adults plus immatures)**	**Number from overseas populations (adults plus immatures)**	**Number from UK population (adults plus immatures)**
NW North Sea	1,523	365	1,158
West of Scotland	861	195	666
SW North Sea	10,177	9,398	779
NW England & Wales	1,657	1,271	386
SW England & Channel	1,153	850	303
Migration BDMPS (September-November and February-April)			
UK North Sea	13,277	10,623	2,654
UK Western waters plus Channel	4,373	2,752	1,621

Colour coding is green for UK numbers and totals because UK breeding numbers have been counted several times in recent decades and are considered to be well known and moderately stable, while wintering numbers off UK coasts have also been surveyed and because red-throated divers tend to occur relatively close to shore their numbers are easier to survey at sea than for species dispersed over larger areas. Numbers from overseas populations are less certain (classified amber except for SW North Sea) but since totals at sea are moderately well known and breeding numbers are well known, numbers from overseas can be assessed against those numbers. Numbers from overseas in the SW North

Sea are thought to be rather well known based on surveys of coastal waters in the southern North Sea which indicate large totals in a region where relatively few UK adults overwinter, and so it can be inferred (supported by ring recovery data) that those birds are predominantly from the Fennoscandian population.

Colour coding is amber for migration numbers, as the numbers and distribution during migration are less well known than for mid-winter, and the migration routes used are only broadly known from the relatively limited ring recovery data for this species. However, colour coding is amber rather than red because population sizes and breeding distributions are well known and largely stable, and the available evidence indicates consistent numbers and migrations from year to year with evidence for birds consistently returning to the same sites by the same routes in successive years, but for immature birds to migrate further south than adults.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 1 to 7.

3.1 Breeding range and taxa

This circumpolar species is monotypic, with Scotland at the southern edge of its breeding range. There appears to be little information about use of biometrics to identify origins of individuals.

3.2 Non-breeding component of the population

Red-throated divers start to breed when 3 years old (BTO Birdfacts). Adult survival rate is 0.84 (BTO Birdfacts), juvenile survival estimated at 0.61 (BTO Birdfacts) and mean productivity is 0.635 chicks per pair (JNCC database, n=136 measurements). To obtain a stable population, survival of immatures was adjusted to 0.72 for juveniles, 0.84 for older age classes. The model population comprised 60% adults, 19% juveniles, 11% 1-year olds, and 10% 2-year olds. There are 0.74 immatures per adult.

3.3 Phenology

Red-throated diver breeding season ends by September-October (Forrester et al. 2007), or the end of September (Pennington et al. 2004), but most birds have left their breeding sites by August-September (Forrester et al. 2007) or mid-August (Pennington et al. 2004). Autumn migration starts in August (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007) or mid-August (Cramp et al. 1977-94). Peak autumn migration occurs in September in Shetland (Pennington et al. 2004), September-October in the UK (Wernham et al. 2002; Forrester et al. 2007), September-November in English waters (Brown and Grice 2005), or October-November in Belgium (Vanermen et al. 2013) or throughout Europe (Cramp et al. 1977-94). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in September-December (Figure 3.1). Autumn migration is completed by November (Pennington et al. 2004; Forrester et al. 2007) or mid-December (Cramp et al. 1977-94). Spring migration starts in February (Pennington et al. 2004; Forrester et al. 2007) or early March (Cramp et al. 1977-94) or March (Wernham et al. 2002). Peak spring migration occurs in February-April in Belgium (Vanermen et al. 2013), in late February and early March in English waters (Brown and Grice 2005), in April (Cramp et al. 1977-94), or in April-May (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Peak rate of change in numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in February-March (Figure 3.1). Spring migration is completed by June (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007) or mid-June (Cramp et al. 1977-94). The first spring records of red-throated diver in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly from 1 January and the last records were predominantly at 31 December, as large numbers of red-throated divers overwinter, while

peak autumn migration was reported in September or October in most years, and peak spring migration was reported in March, April or May in most years. Birds reoccupy nest sites from as early as February, but most return to breeding sites in the UK in mid-March (Pennington et al. 2004; Forrester et al. 2007). Orkney and Shetland Bird Reports indicate modal return to nest sites in February (4 cases), and March (5 cases).

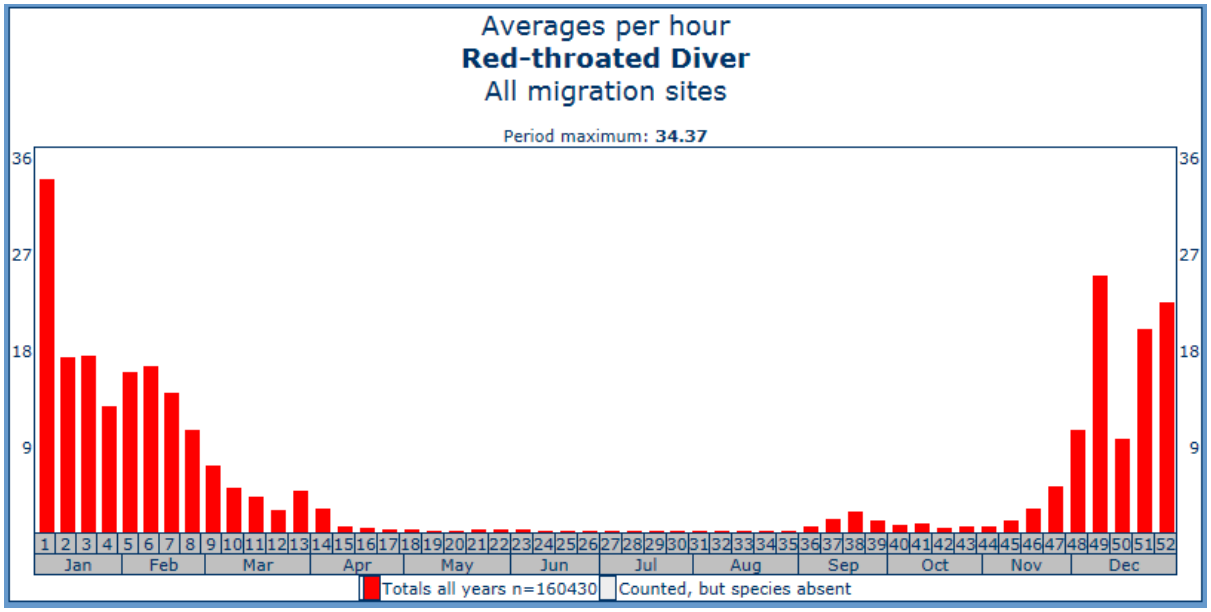


Figure 3.1. Average numbers of red-throated divers counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

3.4 Defined seasons

- UK Breeding season — March-August
  - Migration-free breeding season — May-August
- Non-breeding season
  - **Post-breeding migration in UK waters (migration BDMPS**[1]**)** — **September-November**
  - **Migration-free winter season BDMPS)** — **December-January (winter**
  - **Return migration through UK waters BDMPS**) — **February-April (migration**

Apart from the breeding season, two seasonal BDMPS periods are considered to be appropriate for red-throated diver:

Migration seasons BDMPS (September-November and February-April); and

Winter BDMPS (December-January).

3.5 Movements of birds from the UK population

Red-throated divers breed at freshwater pools close to the coast, but feed in the sea on small fish, and winter inshore on sheltered coasts. The young make their first flight to the sea attended by their parents and then move away from the breeding areas within a few days (Wernham et al. 2002). Scandinavian birds winter in the southern North Sea and southwards

[1] Seasons for which BDMPS have been generated are annotated (BDMPS).

to central France (Eriksson 2000). The Scottish population (of about 1,255 pairs; Gibbons et al. 1997; Dillon et al. 2009) travels shorter distances to winter than more northerly birds (Wernham et al. 2002). Recoveries of birds ringed in Orkney and Shetland show a southerly movement in autumn. Juveniles move furthest, reaching as far south as northwest France (Okill 1994). Adults from Shetland mostly overwinter along Scottish coasts, with some remaining in Shetland (Okill 1994; Wernham et al. 2002). One quarter of one-year olds return to natal areas in their first summer while three quarters remain in wintering areas (Okill 1994). Among two year olds, two-thirds return to natal areas in summer but about one-third remain along northern Scottish coasts, whereas by their third summer all birds return in summer to their breeding area (Wernham et al. 2002).

3.6 Movements of birds from overseas into UK waters

Foreign-ringed birds found in Britain in winter originated from Greenland (3), Finland (4), and Sweden (3) (Wernham et al. 2002). Most were recovered in south-east England (in contrast to most Scottish birds being recovered on Scottish or Irish coasts) (compare Figures 4 and 5 in the chapter on red-throated diver in Wernham et al. 2002). There is no evidence to suggest that red-throated divers from the Russian population (which winters in the Baltic Sea) ever reach the UK (Wernham et al. 2002). No red-throated divers ringed in Iceland (where there are about 1,500 breeding pairs; Hagemeijer and Blair 1997) have been recovered in Britain or Ireland (Wernham et al. 2002), but one Icelandic bird was recovered in the Faroes in its first winter (Hammer et al. 2013). This suggests that Icelandic red-throated divers probably mostly remain in Icelandic waters throughout the year, a suggestion supported by recent geolocator deployments on red-throated divers breeding in Iceland (Ib Krag Petersen pers. comm.). That would make red-throated diver an example of 'leap-frog migration' with birds from the Greenland population migrating past the relatively sedentary populations of Iceland, Faroes and Scotland. Winter populations in Scottish waters seem most likely to be predominantly birds from the Scottish population (and especially adults from that population), with a minority coming from Greenland (where there are about 1,000 pairs; Wetlands International 2006) and Fennoscandia (where there are about 5,500 pairs; Hagemeijer and Blair 1997), and possibly a few from Iceland. Birds wintering in English waters apparently include birds from Scotland (with a high proportion of those being juveniles and immatures rather than adults), Greenland, and Fennoscandia, possibly including small numbers from Iceland. There are only 25 pairs breeding in the Faroes (Hammer et al. 2013) so that population is very small and may well remain in Faroese waters or migrate to Scottish waters, but no birds have been ringed there.

3.7 Numbers in UK waters

O'Brien et al. (2008) estimated that 17,000 red-throated divers overwinter in Great Britain, updating previous estimates from Lack (1986) and Batten et al. (1990) that were underestimates due to lack of knowledge of numbers in the Outer Thames in particular. Of these, just over 10,000 winter between Flamborough Head and Dungeness. In Scottish territorial waters, there were 2,270 in winter, mostly inshore and with larger numbers on the east coast than on the west coast. The UK summer population is estimated to total 4,146 birds (Dillon et al. 2009), and most of these overwinter in British waters (Okill 1994), with adults predominantly in Scottish waters and immatures often further south. This suggests that most of the red-throated divers wintering in Scottish waters are likely to be from the UK population if the estimated numbers present in winter are moderately accurate. In contrast, the much larger numbers overwintering off south-east England could only be explained by presence of large numbers from overseas populations. Given evidence from ring recoveries, these appear to be predominantly birds from Fennoscandia, plus substantial numbers from Greenland. Based on population size it seems likely that no more than about 2,000 of these birds in English waters originate from Scottish breeding areas, whereas about 12,000 are probably from Fennoscandia and Greenland. About 48,000 red-throated divers winter in the area from the Kattegat to the River Elbe, about 43,000 in the Baltic Sea (Danielsen et al.

1993; Brown and Grice 2005). In English waters, wintering red-throated divers are scarce off SW England, uncommon off the south coast, present in large numbers off NW England, but in highest numbers off E England (Brown and Grice 2005).

3.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of Europe, comprising 7,158 pairs, and the biogeographic winter population of Europe as 75,000 individuals (based on data in Rose and Scott 1997), but updated to an estimate of 100,000 to 1,000,000 by Delaney and Scott 2002 (see also Musgrove et al. 2011). Red-throated divers in UK waters originate almost entirely from UK, Fennoscandia or Greenland, so populations outside those areas can be discounted as not occurring in UK waters (e.g. Iceland, Russia), or too small to be relevant (e.g. Faroe), or both. Thus a limited biogeographic population could be defined as birds from UK (1,255 pairs), Greenland (1,000 pairs), and Fennoscandia (5,500 pairs), a total of 7,755 pairs. This is equivalent to a total of 15,500 breeding adults and an associated 11,500 immatures, so a total of 27,000 birds. BirdLife International (2004) suggests a population of 5,000 to 30,000 pairs in Greenland, but this number, which is not supported by any original reference, seems highly unlikely given that previous estimates for Greenland were all around 1,000 pairs.

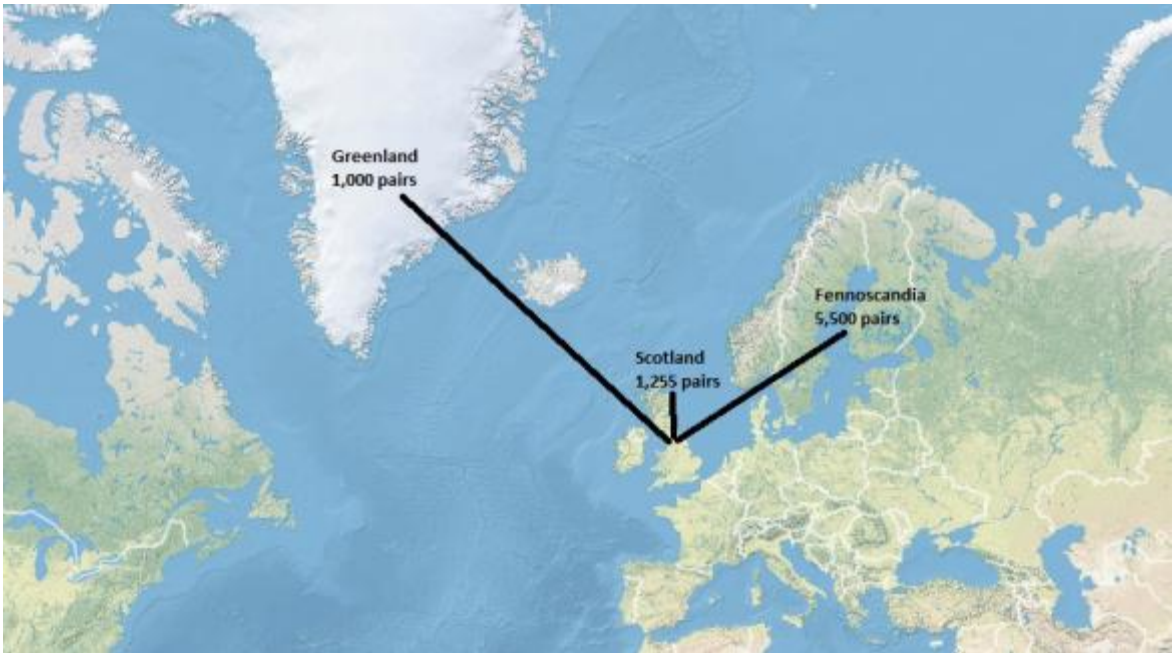


Figure 3.2. Breeding population origins of red-throated divers in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap ©OpenStreetMap contributors.

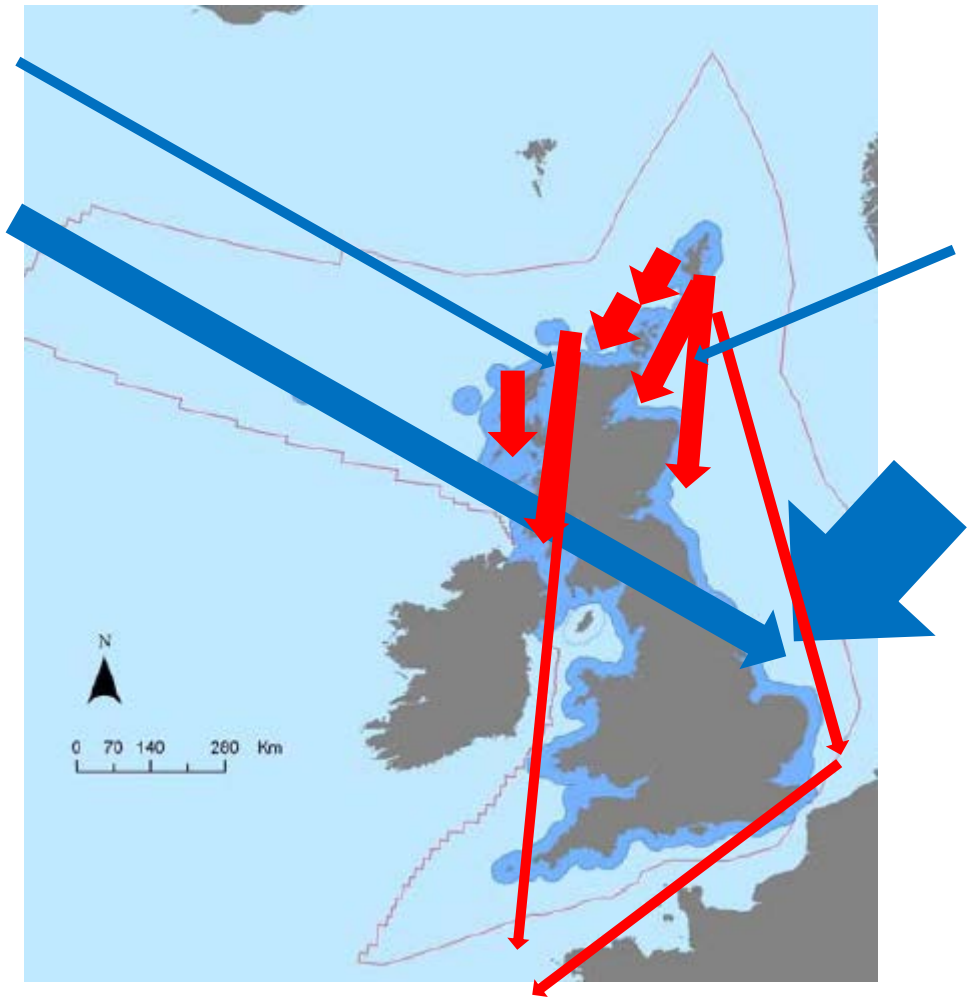


Figure 3.3. Main movements of red-throated divers from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure. Counts of numbers of red-throated divers in winter in different areas around the UK are thought to be moderately accurate (although this represents a change from the past as large numbers have been 'discovered' in recent years in some areas). Those counts, combined with knowledge of movements from ringing studies, give moderate confidence in the fact that relatively few birds from overseas winter in Scottish waters, and that most birds wintering in English waters of the southern North Sea originate from Fennoscandia.

3.9 Proportion of UK population from UK breeding SPAs

The 10 SPAs with breeding red-throated divers as a feature together held 395 pairs at designation, estimated to represent 31% of the UK breeding population of 1,255 pairs (Stroud et al. 2001). Breeding numbers at UK SPAs appear to have generally remained stable (Table 3.1). Breeding numbers in the UK in total also appear to have remained approximately stable over recent decades or increased slightly (Gibbons et al. 1997; Stone et al. 1997; BirdLife International 2004; Baker et al. 2006; Forrester et al. 2007; O'Brien et al. 2008; Musgrove et al. 2013). UK SPAs with red-throated diver as a breeding feature are distributed predominantly in Shetland, Orkney, Caithness, and the western islands of

Scotland (Western Isles and Inner Hebrides) (Figure 3.4). Seven of the SPA populations lie within the NW North Sea region, and three lie in the West of Scotland region (Table 3.1). The SPA populations in the NW North Sea region held a total of 237 pairs in the most recent census at each SPA (Table 3.1). The SPA populations in the West of Scotland region held a total of 108 pairs in the most recent census at each SPA (Table 3.1). It is therefore likely that SPA populations now represent about 27% of the UK Breeding population based on these data. Stroud et al. (2014) estimated that SPA populations represented 30.5% of the GB population in 2006.

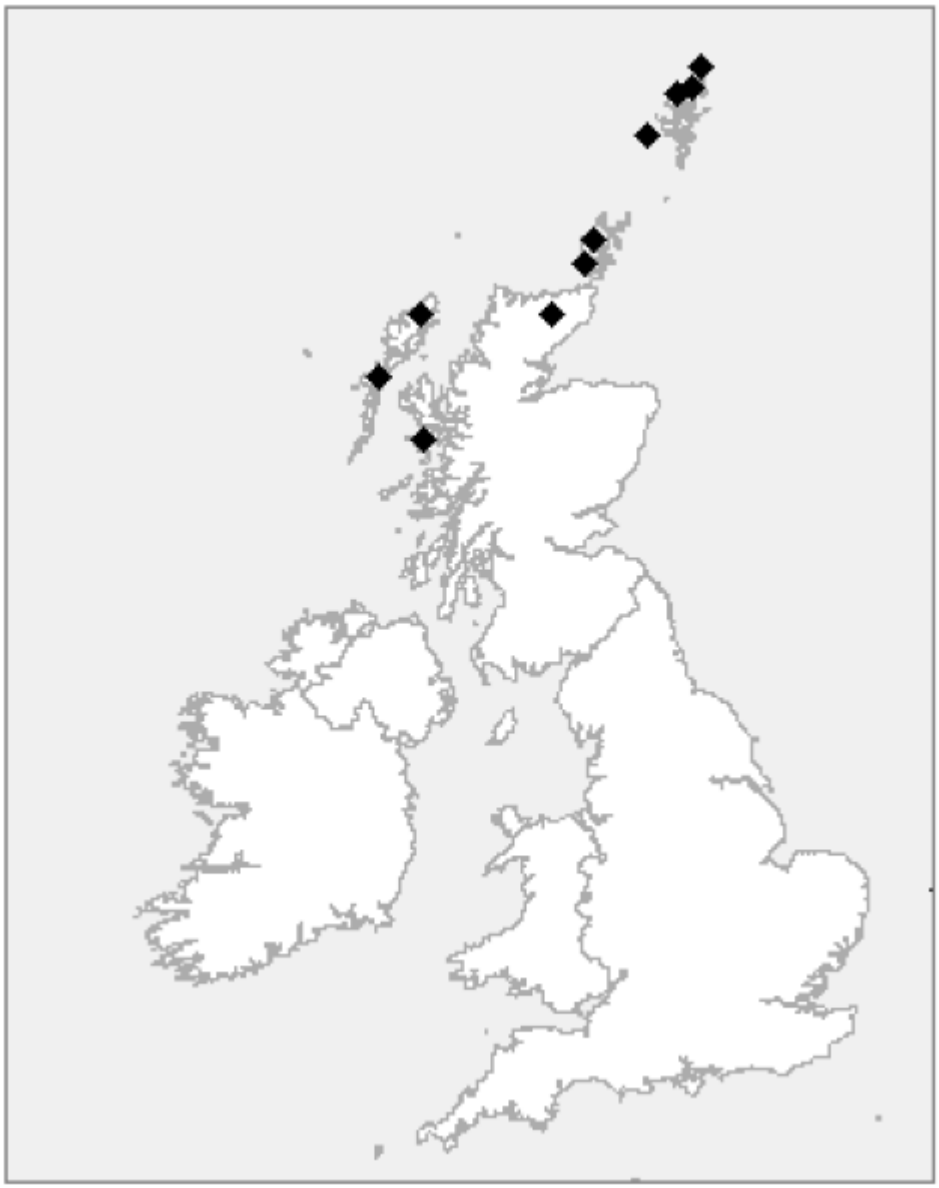


Figure 3.4. Locations of the 10 UK SPAs with red-throated diver as a breeding feature. These SPA populations are listed in Table 3.1. From Stroud et al. 1990.

Table 3.1. The UK SPA suite for breeding red-throated diver.

SPA	Location	Pairs	Year desig-nated	Site condition monitoring*	Latest counts (pairs)	Year	Reference
NW North Sea							
Hermaness, Saxavord & Valla	Shetland NE	28 (1994-1996)	1994	Declined 2013	16	2013	SNH (Bob Bryson in litt.)
Otterswick and Graveland	Shetland NE	27 (1992-1996)	2001	Maintained 2006	>25	2006	SNH (Bob Bryson in litt.)
Ronas Hill North Roe Tingon	Shetland NE	50 (1994)	1997	Maintained 2006	50	2006	SNH (Bob Bryson in litt.)
Foula	Shetland NE	11 (1994)	1995	Maintained 2013	10 12	2012 2013	Gear 2012 Gear 2013
Orkney Mainland Moors	Orkney NE	15 (1994-1996)	2000	Maintained 2007	>28	2007	SNH (Bob Bryson in litt.)
Hoy	Orkney NE	56 (1994)	2000	Maintained 2007	60	2007	SNH (Bob Bryson in litt.)
Caithness & Sutherland Peatlands	N Scotland NE	89 (1993-1994)	1999	Maintained 2006	46	2006	Stroud et al. 2014
West of Scotland							
Lewis Peatlands	Western Isles NW	60 (mid-1990s)	2000	Declined 2004	80	2006	Stroud et al. 2014
Mointeach Scadabhaigh	Western Isles NW	48 (1994)	1999	Maintained 2004	33-35 17	2004 2006	SNH (Bob Bryson in litt.) Stroud et al. 2014
Rum	Inner Hebrides NW	11 (1992-1996)	1982	Maintained 2007	11	2013	SNH (Bob Bryson in litt.)

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

3.10 BDMPS

There are thought to be about 15,300 birds in UK waters in winter, with most birds from the UK population included (about 3,300 birds, allowing for small numbers of immatures wintering further south), and about 12,000 birds from the overseas populations. During migration, there will be marginally larger numbers passing through UK waters as some birds winter further south in Europe; during migration around 17,300 birds, about 4,300 from the UK and about 13,000 from overseas populations.

It makes biological sense to consider Scottish North Sea waters separately from English North Sea waters, since it seems that most birds wintering in Scottish North Sea waters are from the UK population, whereas most birds wintering in English North Sea waters are from Fennoscandia. It also makes sense to separate the populations to the west and east of mainland UK. Most red-throated divers from SPA populations in the Western Isles and Inner Hebrides winter to the west of the UK mainland, whereas probably most of those from SPA populations in the NW North Sea winter in the North Sea. Red-throated divers wintering off NW England may be a mixture of birds from UK populations and from Greenland. Only small numbers winter in the English Channel and SW England, but probably include a mixture of

mainly immatures from the UK population and birds from Greenland and Fennoscandia. Therefore, proposed BDMPS regions are as shown in Figure 3.5.

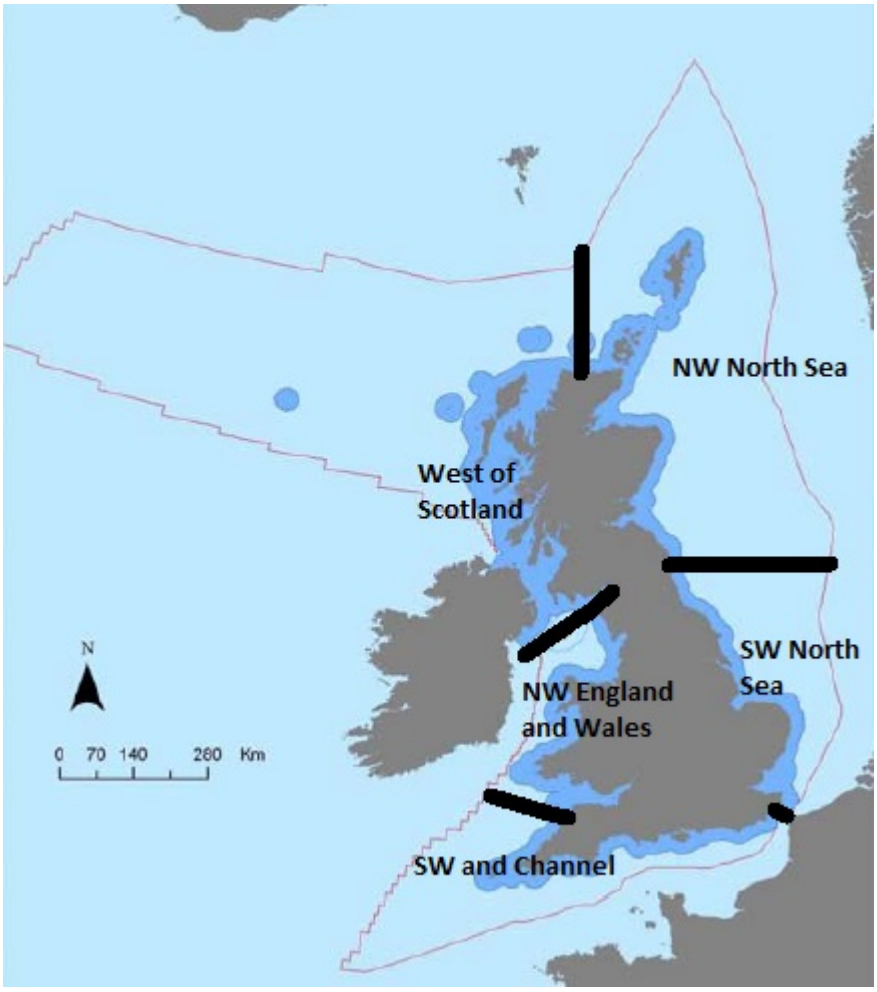


Figure 3.5. Five defined BDMPS spatial areas for red-throated divers in UK waters in winter. Limits of UK waters are shown by red line. BDMPS spatial areas extend from the UK coast to the red limit, bounded by the thick black lines marking the sides of each BDMPS area. The five BDMPS are 'NW North Sea', 'SW North Sea', West of Scotland', NW England & Wales' and 'SW and Channel'. For migration seasons there are two BDMPS, 'North Sea' (NW and SW North Sea combined) and 'Western waters plus Channel' (West of Scotland, NW England & Wales, and SW & Channel areas combined).

It is estimated that about 50% of adults from breeding areas in the northern isles and North Sea coast of Scotland winter in the NW North Sea whereas only a very few (perhaps 5%) of those from western UK breeding areas winter in the NW North Sea (based on literature reviewed in section 3.5). Very few birds from Greenland or Fennoscandia have been recovered in the NW North Sea (section 3.6) but it seems likely that some birds from Greenland will stop in the NW North Sea rather than continuing to the SW North Sea so the proportions wintering in this area are estimated at 5% and 1% respectively. There is apparently very little movement of adults from western breeding areas to winter in the NW North Sea (section 3.5), so this proportion is estimated at 5%. The same percentages are applied for immature birds from western UK, Greenland and Fennoscandia as for adults. There is evidence for birds from the northern isles that many immatures winter further south so the proportion of those in the NW North Sea in winter is estimated at 20% with most moving further south. These combinations of proportions result in an estimated winter BDMPS in NW North Sea of 1,523 birds, a number that is consistent with the counts of red-

throated divers wintering in NW North Sea (Section 3.7). Details of apportioning and estimated numbers are in Appendix A Table 1.

It is estimated that about 20% of adults from breeding areas in the northern isles and North Sea coast of Scotland winter in the SW North Sea whereas only a very few (perhaps 5%) of those from western UK breeding areas winter in the SW North Sea (based on literature reviewed in section 3.5). Ringed birds from Greenland and Fennoscandia have been recovered in the SW North Sea (section 3.6) and the numbers in that area in winter require a substantial movement of birds from those populations to winter there. Based on the observation that divers are likely to avoid migrating overland, it seems likely that birds from Greenland make up a higher proportion of the overseas birds wintering in UK western waters and that most overseas birds in North Sea waters originate from Fennoscandia rather than Greenland, so the proportions wintering in this area are estimated at 2% and 40% respectively. Similarly, there is apparently very little movement of adults from western breeding areas to winter in the SW North Sea (section 3.5), so this proportion is estimated at 5%. The same percentages are applied for immature birds from western UK. For Fennoscandia it is estimated that 60% of immatures winter in the SW North Sea because it is generally the case that immature red-throated divers winter further south than adults. For Greenland it is estimated that 5% of immatures winter in the SW North Sea because it is generally the case that immature red-throated divers winter further south than adults. There is evidence for birds from the northern isles that many immatures w*i*nter further south so the proportion of those in the SW North Sea in winter is estimated at 30% for immatures compared to 20% for adults. These combinations of proportions result in an estimated winter BDMPS in SW North Sea of 10,177 birds, a number that is consistent with the counts of red-throated divers wintering in SW North Sea (Section 3.7). Details of apportioning and estimated numbers are in Appendix A Table 2.

It is estimated that about 5% of adults from breeding areas in the northern isles and North Sea coast of Scotland winter in West of Scotland waters whereas 40% of adults from western UK breeding areas winter in West of Scotland waters (based on literature reviewed in section 3.5). There is evidence for birds from the northern isles that many immatures w*i*nter further from the breeding area so the proportion of those in West of Scotland waters in winter is estimated at 20% for immatures compared to 40% for adults for birds from western populations. Immatures from North Sea UK populations are likely to be more represented in west of Scotland waters than adults, so the proportion is estimated at 10% for immatures compared to 5% for adults. Based on the observation that divers are likely to avoid migrating overland, it seems likely that birds from Greenland make up a small proportion of the overseas birds wintering in UK western waters, so the proportions wintering in this area are estimated at 2% and 5% respectively for adults and immatures. There is no evidence from ringing that birds from Fennoscandia winter west of Scotland, so proportions from that population are set at zero for adults but 1% for immatures. That is also consistent with total numbers wintering west of Scotland being relatively small, and can be accounted for by the proportions estimated above. These combinations of proportions result in an estimated winter BDMPS for the West of Scotland area of 861 birds, a number that is consistent with the counts of red-throated divers wintering in the West of Scotland area (Section 3.7). Details of apportioning and estimated numbers are in Appendix A Table 3.

It is estimated that about 2% of adults from breeding areas in the northern isles and North Sea coast of Scotland winter in NW England and Wales waters whereas 20% of adults and immatures from western UK breeding areas winter in NW England and Wales waters (based on literature reviewed in section 3.5). Immatures from North Sea UK populations are likely to be more represented in NW England and Wales waters than adults, so the proportion is estimated at 5% for immatures compared to 2% for adults. Based on the observation that divers are likely to avoid migrating overland, it seems likely that birds from Greenland make up a small proportion of the overseas birds wintering in NW England and Wales waters, so

the proportions wintering in this area are estimated at 10% and 30% respectively for adults and immatures. Probably few birds from Fennoscandia winter in NW England and Wales (but there is one ring recovery), so proportions from that population are set at 2% for adults but 5% for immatures. These combinations of proportions result in an estimated winter BDMPS for the NW England and Wales area of 1,657 birds, a number that is consistent with the counts of red-throated divers wintering in the NW England and Wales area (Section 3.7). Details of apportioning and estimated numbers are in Appendix A Table 4.

It is estimated that about 2% of adults and 5% of immatures from breeding areas in the northern isles and North Sea coast of Scotland winter in SW England and Channel waters whereas 10% of adults and 20% of immatures from western UK breeding areas winter in SW England and Channel waters (based on literature reviewed in section 3.5). Immatures from North Sea UK populations are likely to be more represented in SW England and Channel waters than adults, so the proportion is estimated at 5% for immatures compared to 2% for adults. Based on the observation that divers are likely to avoid migrating overland, it seems likely that birds from Greenland make up a proportion of the overseas birds wintering in SW England and Channel waters, so the proportions wintering in this area are estimated at 10% and 20% respectively for adults and immatures. Probably few birds from Fennoscandia winter in SW England and Channel waters (but there is one ring recovery in the area), so proportions from that population are set at 1% for adults but 3% for immatures. These combinations of proportions result in an estimated winter BDMPS for the SW England and Channel waters of 1,153 birds, a number that is consistent with the counts of red-throated divers wintering in SW England and Channel waters (Section 3.7). Details of apportioning and estimated numbers are in Appendix A Table 5.

Ringing data indicate that most red-throated divers from the northern isles move southwards in autumn through the North Sea and that birds from western UK populations mostly move south through western waters (Section 3.5) but there is some evidence from ringing of small numbers moving between western waters and North Sea and vice versa. In computing BDMPS for these two areas for the migration seasons it is therefore estimated that 95% of adults and 80% of immatures from UK North Sea populations are in UK North Sea waters (NW plus SW North Sea areas) during migration seasons (September-November and February-April), while possibly 5% of western UK red-throated divers (adults and immatures) also pass through North Sea waters on migration. Similarly it is estimated that 95% of adults and 80% of immatures from western populations migrate through western waters (West of Scotland to Channel) while 5% of North Sea adults and 20% of North Sea immatures (birds from the northern isles) migrate through western waters. Large numbers from Fennoscandia migrate through the North Sea but ring recoveries indicate that few reach western waters, so proportions estimated for this population are 45% of adults and 65% of immatures migrating through UK North Sea waters, with 5% and 10% respectively in western waters. Conversely it seems likely that birds from Greenland migrate more through western waters than through the North Sea, so proportions were estimated at 8% of adults and 15% of immatures migrating through UK North Sea waters and 25% of adults and 60% of immatures through western waters. These percentages result in estimated numbers in the migration season BDMPS that are consistent with diver count data and estimates in the literature (Section 3.7). Details of apportioning and estimated numbers are in Appendix A Tables 6 and 7.

3.11 Proportion of UK SPA birds in each BDMPS

Proportions of each BDMPS that are adults from UK SPA breeding populations can be calculated directly from Appendix A Tables 1 to 7. For example, in the UK NW North Sea area in winter (Appendix A Table 1) there are 248 adults from SPA populations in the winter BDMPS of 1,523 birds, so approximately 16% of birds in that BDMPS are adults from SPA populations. In contrast, for the UK SW North Sea area in winter (Appendix A Table 2) there are 105.6 adults from SPA populations in the winter BDMPS of 10,177 birds, so

approximately 1% of birds in that BDMPS are adults from SPA breeding populations (SPA populations for wintering birds are not considered in this calculation).

3.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Red-throated divers migrate primarily through coastal waters, and winter in shallow coastal waters. Their distribution across the regions will therefore be far from uniform, with almost all birds close to the coast and predominantly in more sheltered areas. Birds from SPA populations may tend to winter relatively close to their SPA breeding sites, but this is uncertain. Given that the spatial distribution of SPAs is similar to the spatial distribution of the broader breeding population of the species in Scotland, it is likely that the proportion of birds from SPAs will be fairly consistent throughout Scottish waters. In England, it is likely that a high proportion of the birds from Scottish SPAs will be immatures rather than breeding adults, since the immatures winter further south than adults. However, most birds in southern North Sea waters are likely to be from Fennoscandia rather than the UK population, and birds from UK SPA populations are likely to be fairly randomly distributed amongst these.

4. GREAT NORTHERN DIVER *Gavia immer*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (September to May) (adults and immatures)
Overseas	430,000	4,000
UK	0	0
Total	430,000	4,000

Non-breeding season BDMPS (September to May)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK breeding population (adults plus immatures)
West of Scotland	2,000	2,000	0
NW North Sea	1,000	1,000	0
SW North Sea & Channel	200	200	0
NW England & Wales	300	300	0
SW England	500	500	0

Breeding numbers in overseas populations are not well known so are coded red. Colour coding is green for numbers from UK breeding population because great northern divers do not normally breed in Britain. Colour coding is amber for numbers of birds from overseas and in total because the species is not easy to count at sea, but there have been dedicated surveys of wintering divers in UK waters that appear to provide moderately accurate numbers in each region. Great northern divers are apparently highly faithful to the same wintering site in successive years and numbers appear to be fairly stable across years. There is, however, a possibility that wintering numbers are higher than counts indicate (for example numbers oiled in Shetland in one oil spill exceeded the numbers thought at the time to be present). There is also some uncertainty about numbers migrating through UK waters, although those numbers are likely to be similar to the wintering numbers as relatively few great northern divers winter further south in Europe than UK waters. Migration routes are also uncertain, but it seems likely that birds arrive directly at, and depart directly from, winter areas rather than necessarily moving northwards through UK waters, since their breeding sites lie far to the west or north-west and migrations must involve long trans-Atlantic flights. Origins of birds from overseas in UK waters have been quite well established from biometric analysis. Because there is no clear evidence for numbers migrating through UK waters being significantly different from numbers wintering in UK waters, a single BDMPS has been defined for the non-breeding period (September to May). If knowledge of migrating numbers improves in future there might be merit in separating this into seasonal BDMPS for migration seasons and for winter.

4.1 Breeding range and taxa

Great northern diver is a monotypic species with a predominantly Nearctic breeding range, from Alaska to Greenland and Iceland, where it nests at large freshwater lakes. Although monotypic, there is variation among populations in biometrics which can be used to identify origins of individuals (Weir et al. 1996).

4.2 Non-breeding component of the population

Great northern divers start to breed when 6 years old (BTO Birdfacts). Adult survival rate is unknown (BTO Birdfacts), juvenile survival unknown (BTO Birdfacts) and mean productivity is unknown. Assuming an adult survival rate of 0.9 (typical of birds recruiting when 6 years old) and productivity of 0.635 chicks per pair (as in red-throated diver), to obtain a stable population, survival of immatures was adjusted to 0.7 for juveniles, 0.8 for 1 and 2 year olds, 0.88 for 3 year olds and 0.9 for older age classes. The model population comprised 48% adults, 15% juveniles and 37% older immatures. There are 1.1 immatures per adult.

4.3 Phenology

Autumn migration starts in August (Wernham et al. 2002), late-August (Cramp et al. 1977-94), September (Forrester et al. 2007) or late September (Pennington et al. 2004). Peak autumn migration occurs in late October in Shetland (Pennington et al. 2004) and in English waters (Brown and Grice 2005), October-November in the UK (Wernham et al. 2002; Forrester et al. 2007), or throughout Europe (Cramp et al. 1977-94). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in September-November (Figure 4.1). Autumn migration is completed by December (Pennington et al. 2004; Forrester et al. 2007) or late December (Cramp et al. 1977-94).

Spring migration starts in early March (Cramp et al. 1977-94), March (Pennington et al. 2004; Forrester et al. 2007) or late April (Wernham et al. 2002). Peak spring migration occurs in early April from English waters (Brown and Grice 2005), in April (Cramp et al. 1977-94), in April-May (Forrester et al. 2007), or May (Wernham et al. 2002; Pennington et al. 2007). Peak rate of change in numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in February-March but with a smaller but distinct peak in May (Figure 4.1). Spring migration is completed by early June (Cramp et al. 1977-94), or June (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007).

The first spring records of red-throated diver in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly from 1 January and the last records were predominantly at 31 December, as large numbers of great northern divers overwinter, while peak autumn migration was reported in October in most years, and peak spring migration was reported in April-May or May in most years.

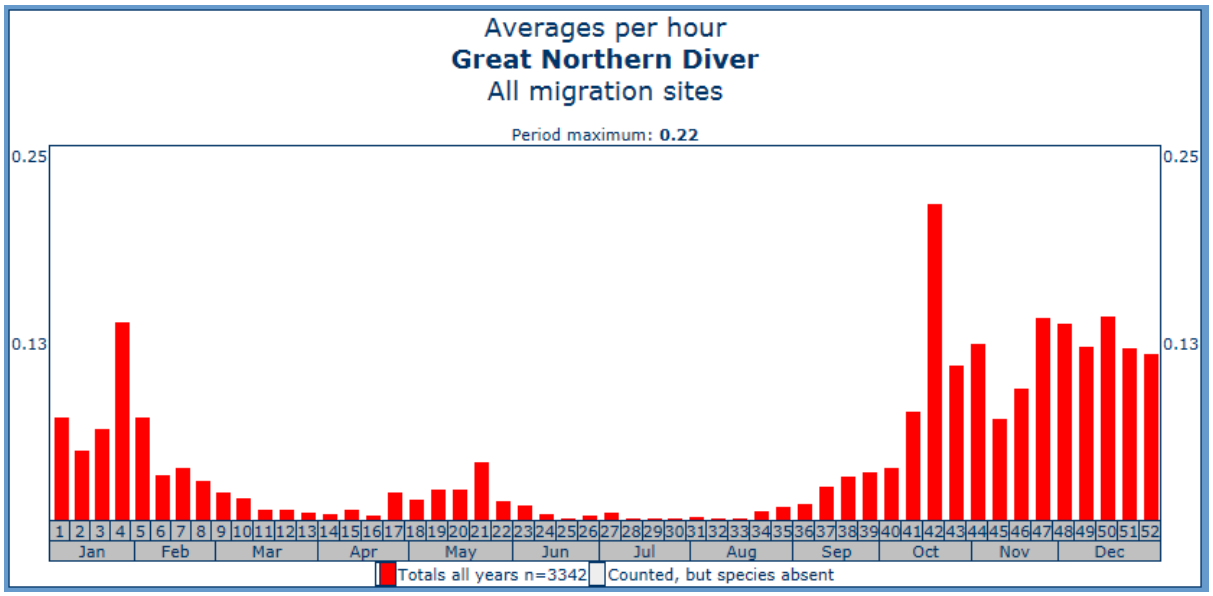


Figure 4.1. Average numbers of great northern divers counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

4.4 Defined seasons:

- UK Breeding season — not applicable
- Post-breeding migration in UK waters — September-November
- **non-breeding season** — **September-May** (**BDMPS**)
- Return migration through UK waters — March-May
- Migration-free breeding season — not applicable
- Migration-free winter season — December-February

Apart from the breeding season, one seasonal BDMPS period was considered to be appropriate for great northern diver:

Non-breeding season (September-May).

4.5 Movements of birds from the UK population

With the very rare exception of the odd pair, or individual, in occasional years, the species does not breed in the UK.

4.6 Movements of birds from overseas into UK waters

Relevant breeding populations are 300-500 pairs in Iceland, 400-1,800 pairs in Greenland, and around 100,000 pairs in Canada (Wernham et al. 2002; Wetlands International 2006). Birds mostly leave breeding areas in September-October, but some arrive in NW Scotland in August. Spring migration occurs in April-May, but substantial numbers of immature birds remain in British waters through the summer. Measurement of great northern diver study skins in the National Museums of Scotland suggested, on the basis of biometric differences between populations, that 45% of those wintering in Scotland were from the Icelandic population, 45% from Greenland and Baffin Island, and only 10% from mainland Canada (Weir et al. 1996). Camphuysen et al. (2010) looked at a sample killed by the Prestige oil spill in Galicia, and concluded that most birds wintering off Spain appear to be juveniles, but that biometrics suggest those birds also come from Iceland and Greenland rather than mainland Canada. Most Canadian birds therefore appear to overwinter in North America rather than migrating to Europe.

4.7 Numbers in UK waters

While many great northern divers winter inshore off coasts of North America, about 6,000 winter inshore from northern Norway to northern Spain (Pennington et al. 2004), of which about 3,500-4,500 individuals winter off Britain and Ireland (Wernham et al. 2002). Wintering birds in British waters are mostly found in shallow sea off the west and north coasts of Scotland and adults seem to predominate in those areas (Weir et al. 1996). Numbers wintering in English waters are unlikely to exceed 1,000 birds, most of which winter off SW England (Brown and Grice 2005). Given the predominance of adults in Scottish waters it is likely that most birds wintering in English waters are immatures (since numbers of immatures are similar to numbers of adults, and in almost all seabirds the immatures winter further from the source population than do the adults).

4.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of Iceland and Greenland, comprising 700-2,300 pairs (BirdLife International 2004 suggests 500 to 2,000 pairs in Greenland but provides no reference to support this), and the biogeographic winter population of Europe as 5,000 individuals (based on data in Rose and Scott 1997). A population of 700-2,300 pairs will have an associated component of immature birds numbering about 1,400 to 4,600 individuals. So the total population size can be estimated at 1,400 to 4,600 birds (Iceland plus Greenland). Since the UK also receives birds from eastern Canada that overwinter in UK waters, it could be appropriate to include that population in the biogeographic population with connectivity to UK waters (and that has been done in this report). However, the Canadian population is very large (perhaps 100,000 pairs) and only a very small proportion of birds wintering in the UK originate from that population (about 10% of birds wintering in UK waters), so it may be appropriate (and precautionary) to omit that population from consideration. The numbers wintering in UK waters (about 3,500 to 4,500 birds) appear to represent the vast majority of the populations from Iceland and Greenland, based on this comparison of breeding numbers, population demography, and wintering numbers.

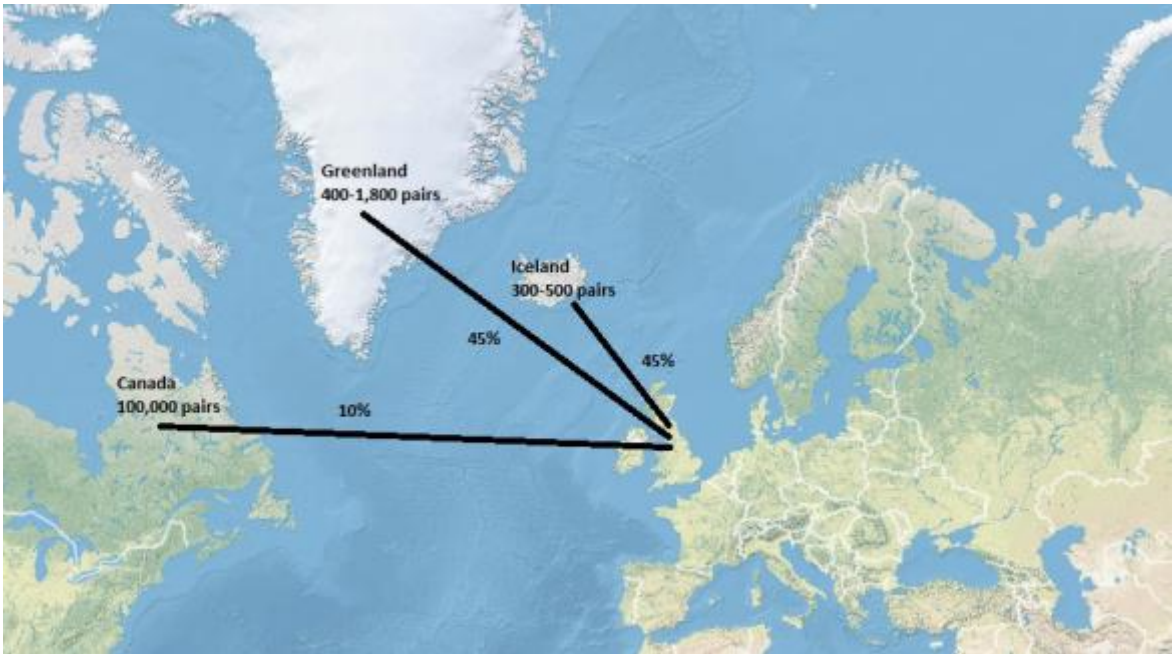


Figure 4.2. Breeding population origins of great northern divers in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given, as are the proportions from each source population represented in non-breeding populations in UK waters. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors

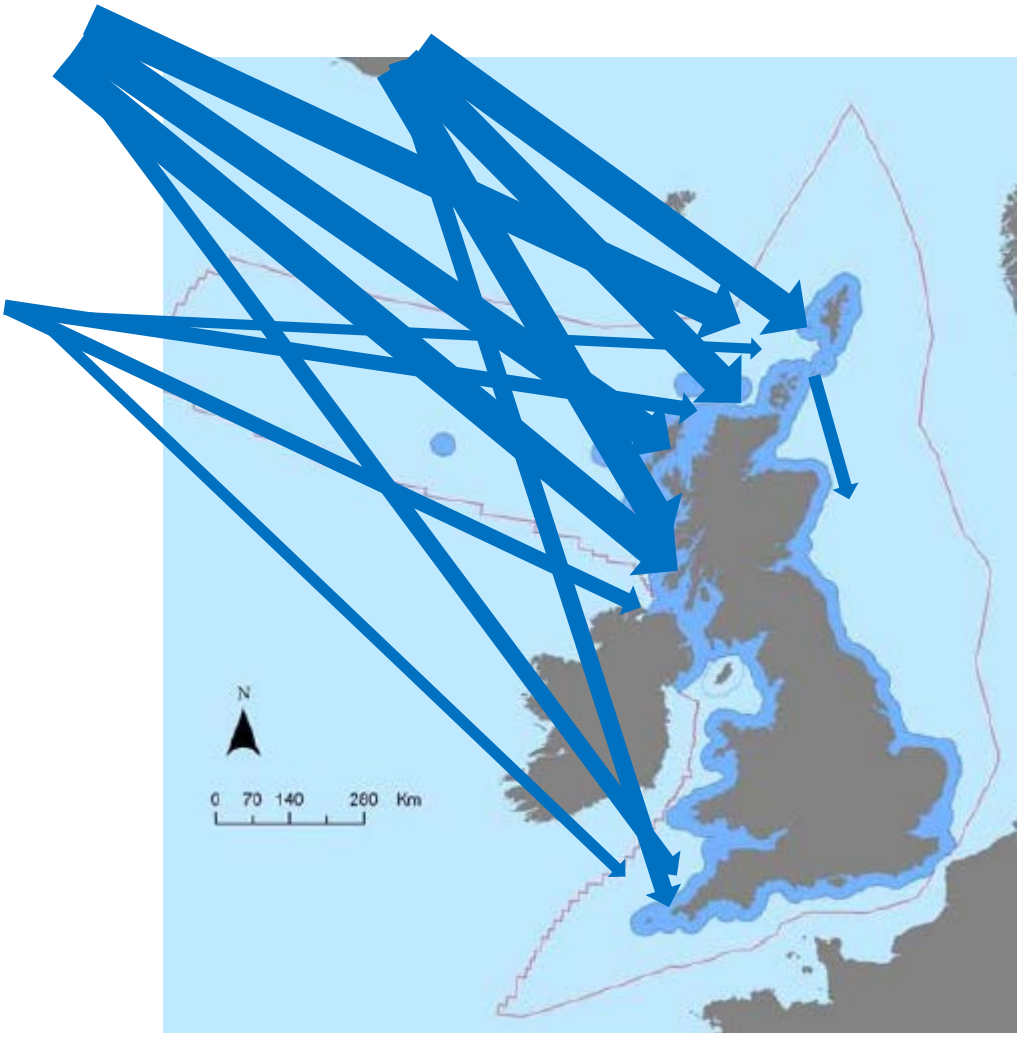


Figure 4.3. Main movements of great northern divers from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure. Museum based studies of biometrics of great northern divers collected from sites in UK waters suggest that the proportions of birds from Iceland (45%), Greenland (45%) and eastern Canada (10%) are consistent across regions, and therefore that there is little or no difference in the use of UK regions between these source populations. The proportions of the source populations wintering in UK waters probably do vary considerably, since the Canadian population is much the largest but represents only 10% of birds wintering in the UK. Wintering numbers are highest in the north and west of the UK.

4.9 Proportion of UK population from UK breeding SPAs

There are no breeding great northern divers in the UK in most years, and no SPAs in the UK include breeding great northern diver as a feature.

4.10 BDMPS

Since great northern divers from all three source populations appear to be similarly represented in different regions, the entire UK waters could be treated as a single BDMPS for this species. However, numbers wintering in different regions are moderately well known,

and are much higher in West of Scotland than elsewhere. Numbers are higher in the NW North Sea than in English waters, where the main concentration of the species is found in SW England. Division into the 5 regions shown in Figure 4.4 may therefore be useful.

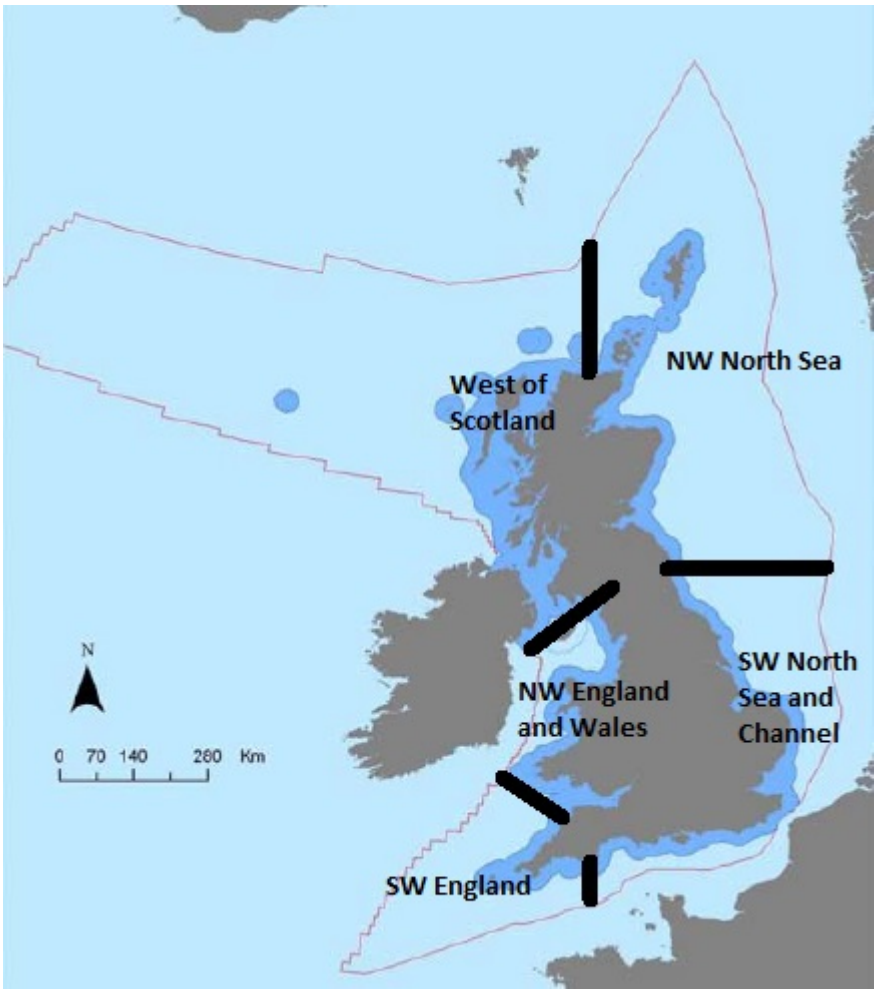


Figure 4.4. Five defined BDMPS spatial areas for great northern divers in UK waters; 'NW North Sea', SW North Sea & Channel', West of Scotland', NW England & Wales', and 'SW England'.

It appears that the proportions from each source population are similar in all the defined regions: 45% from Iceland, 45% from Greenland, and 10% from eastern Canada. About 3,000 of these birds winter in Scottish and Northern Irish waters, with perhaps 2,000 in the West of Scotland region and 1,000 in the NW North Sea region. About 1,000 birds winter in English and Welsh waters, with perhaps 500 of those in the SW England region, 300 in NW England and Wales and 200 in SW North Sea and Channel. Confidence in these numbers is moderate. None of these birds originate from UK breeding SPA populations.

5. NORTHERN FULMAR *Fulmarus glacialis*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in winter (November) (adults and immatures)	Numbers in UK waters in migration seasons (September-October and December-March) (adults and immatures)
Overseas	6,435,000	192,826	385,652
UK	1,620,000	932,277	1,400,044
Total	8,055,000	1,125,103	1,785,696

	Total number of birds in BDMPS (adults and immatures)	Number from overseas populations (adults and immatures)	Number from UK population (adults and immatures)
'Winter' BDMPS (November)			
UK North Sea waters	568,736	96,413	472,323
UK Western waters plus Channel	556,367	96,413	459,954
'Migration seasons' BDMPS (September & October, December to March)			
UK North Sea waters	957,502	192,826	764,676
UK Western waters plus Channel	828,194	192,826	635,368

Colour coding for numbers from overseas populations is red since these overseas populations are very large and while only a very small proportion of those birds pass through or winter in UK waters, this makes estimating numbers very difficult. Although there are ring recovery data, fulmar recoveries provide only a very weak picture of migrations and winter distribution (as with other highly pelagic species), and there are very few tracking studies of this species up until now. Colour coding for UK numbers is amber as these are moderately well documented from breeding colony surveys, but some counts are relatively old (from 1999-2002) and there is evidence for declines in numbers at some colonies though this appears patchy and may partly reflect changes in breeding effort rather than population size.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 8 to 11.

5.1 Breeding range and taxa

The fulmar has a circumpolar breeding range, with two subspecies; *rodgersii* which is found in the northern North Pacific, and nominate *glacialis* which is found in the northern North Atlantic. In the North Atlantic, there are two colour phases of plumage. Birds at colonies at low latitude are all pale phase birds, whereas in the high Arctic most birds are dark phase 'blue' fulmars. Biometrics do not seem to be useful in identifying origins of individuals.

5.2 Non-breeding component of the population

Fulmars start to breed when 9 years old (BTO Birdfacts). Adult survival rate is 0.972 (BTO Birdfacts), juvenile survival unknown (BTO Birdfacts) and mean productivity is 0.424 chicks per pair (JNCC database, n=455 measurements). To obtain a stable population, survival of immatures was adjusted to 0.5 for juveniles, 0.6 for 1-year olds, 0.8 for 2-year olds, 0.9 for 3-6 year olds, 0.92 for 7-year olds and 0.95 for 8-year olds. The model population comprised 62% adults, 13% juveniles and 25% older immatures. There are 0.62 immatures per adult.

5.3 Phenology

The end of the breeding season is described as late August (Forrester et al. 2007) or early September (Pennington et al. 2004). Modal departure from colonies is in August (Pennington et al. 2004; Forrester et al. 2007). However, autumn migration starts in July (Cramp et al. 1977-94), August (Wernham et al. 2002; Forrester et al. 2007) or early September (Pennington et al. 2004). Peak autumn migration occurs in September-October (Cramp et al. 1977-94; Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in September-October but was not clearly pronounced (Figure 5.1). Autumn migration is completed by November (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007) or November-December (Cramp et al. 1977-94).

Spring migration starts in November (Forrester et al. 2007), January (Cramp et al. 1977-94; Pennington et al. 2004) or February (Wernham et al. 2002). Peak spring migration occurs in January-March (Forrester et al. 2007), January-April (Pennington et al. 2004), February-March (Cramp et al. 1977-94), or in March-April (Wernham et al. 2002). Peak rate of change in numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late January-March (Figure 5.1). Spring migration is completed by April (Cramp et al. 1977-94; Forrester et al. 2007) or May (Wernham et al. 2002; Pennington et al. 2004).

Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 reported fulmars present from 1 January to 31 December, but peak autumn migration was reported in September in most years, and peak spring migration was reported in January in most years. The breeding season (birds returning to nest sites) starts from October, but modal return is in November-January (Pennington et al. 2004; Forrester et al. 2007).

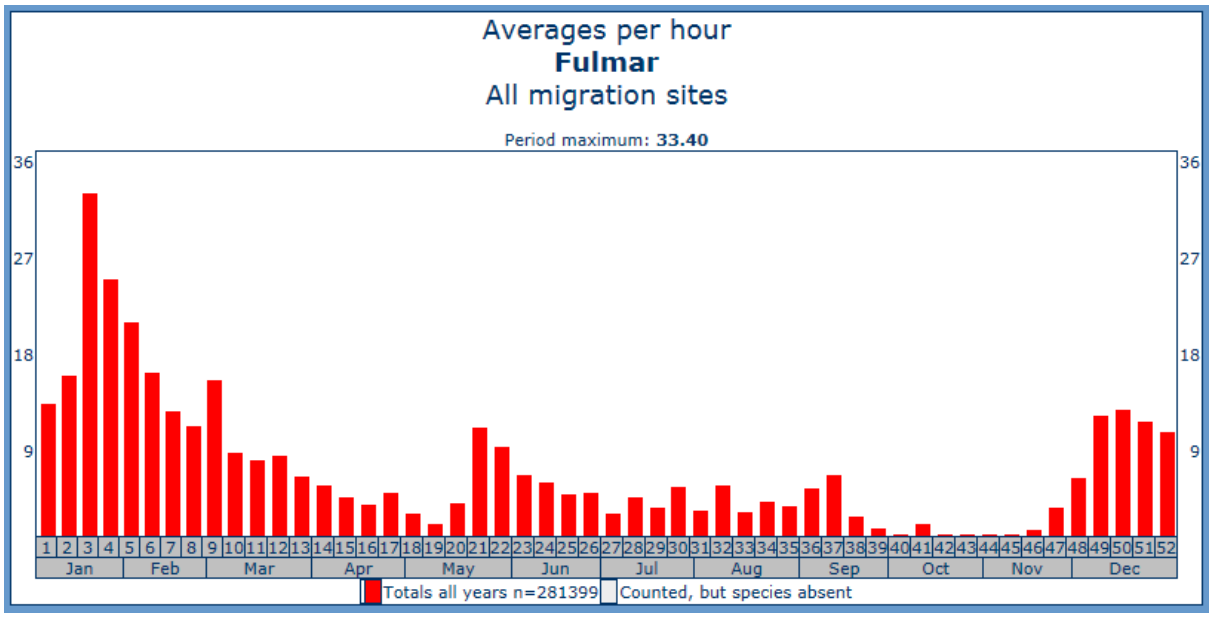


Figure 5.1. Average numbers of fulmars counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as March-July, non-breeding season August-February. However, from the data reviewed above, a more appropriate definition would be breeding season January-August, non-breeding season September-December.

5.4 Defined seasons:

- UK Breeding season — January-August
- Post-breeding migration in UK waters — September-October (**migration BDMPS**)
- non-breeding season — September-December
- Return migration through UK waters — December-March (**migration BDMPS**)
- Migration-free breeding season — April-August
- Migration-free winter season — November (**winter BDMPS**)

Apart from the breeding season, two seasonal BDMPS periods are considered to be appropriate for northern fulmar:

Migration seasons BDMPS (September-October and December-March); and

Winter BDMPS (November).

5.5 Movements of birds from the UK population

After fledging, young fulmars from colonies in the British Isles spend about four years at sea, during which time they disperse widely over the eastern and western North Atlantic, Norwegian and Barents Seas and the Arctic (Macdonald 1977; Wernham et al. 2002). As older immatures, they tend to return to their natal area in summer but for shorter periods than the breeding birds (Forrester et al. 2007). When chicks fledge in August-September, breeders disperse away from the colony and complete moult at sea before returning to re-occupy nest sites only about two to six months later. Breeders attend nest sites from early winter through to chick fledging in August-September. However, fulmars can travel hundreds of kilometres during foraging trips while breeding, and nest site attendance in winter is sporadic so even longer trips may occur at that time of year. Nest attendance in winter

seems to be mainly by males (Wernham et al. 2002) and so females may range over larger areas in winter than males. 'Spring' (i.e. pre-breeding) migration back to colonies must occur in October-February. Many (apparently between 100 and 200) fulmars ringed as chicks at colonies in Britain have been recovered in the Faroes (Hammer et al. 2013). Most (over 80%) of those were deliberately harvested for food, and predominantly caught as immatures. However, recoveries of fulmars ringed at British colonies provides a very incomplete picture of migrations and wintering areas as the chances of ringed birds being recovered are extremely low in many areas such as the mid-Atlantic or high Arctic.

5.6 Movements of birds from overseas into UK waters

Up to publication of the Migration Atlas, 22 foreign-ringed fulmars had been recovered in the British Isles. Seven of these had been ringed at sea so were of uncertain population of origin. The others came from the Faroes, Iceland, Denmark, and Norway (Wernham et al., 2002). Given the very uneven distribution of ringing effort among fulmar populations, these data provide only a very crude indication of the origins of fulmars that are present in British waters (Wernham et al. 2002), but suggest that most may come from Faroes, Iceland, and Norway. Fulmars from high Arctic populations are predominantly of the dark colour morph 'blue fulmars'. These birds are occasionally seen in British waters, especially in winter, but represent a very small proportion of the fulmars present, suggesting that numbers of fulmars from high Arctic populations reaching British waters are negligible and that the vast majority of birds seen in British waters are either from British colonies, or from populations in Faroe, Iceland or Norway. There are around 500,000 pairs in the UK, 600,000 in Faroe, 1.5 million pairs in Iceland (though numbers breeding there declined by 30% from 1983-86 to 2005-08; Gardarsson 2006, Gardarsson et al. 2011), and 386,000 pairs in Norway. All of these populations will have large numbers of immature birds associated with them. The tendency for breeding age birds to attend colonies from October-November through to August-September suggests that most fulmars in British waters are likely to be from UK colonies, but the high numbers in populations in Faroe, Iceland and Norway, together with the relatively mobile nature of immature fulmars, suggests that an unknown but potentially moderately high proportion of birds in British waters could originate from those populations, especially in early winter.

5.7 Numbers in UK waters

ESAS data suggest that there are about 2 to 50 birds per km^2 in Scottish territorial waters in winter (Forrester et al. 2007). From surveys in 2007 and 2008, Fauchald and Tveraa (2009) reported mean densities at sea of 80-400 birds per km^2 in the Norwegian Sea in spring/summer, and 25-300 birds per km^2 in the Barents Sea in autumn, suggesting much higher densities than found around the UK. Only low densities occur in English waters (Stone et al. 1995; Brown and Grice 2005). However, Forrester et al. (2007) suggest that about 1,000,000 fulmars are in Scottish waters during winter (defined in that work as December-February so note that much of that period would involve breeding birds already being back at nest sites). During migration periods, densities of fulmars are higher than in winter, and suggest that closer to 2,000,000 birds are present at sea in UK waters during peak migration seasons, a number that is still only slightly greater than the total population of the UK (including immatures) so does not indicate that there are necessarily large numbers of birds from overseas populations passing through UK waters even during the migration period.

5.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the subspecies *glacialis* which breeds around the North Atlantic, comprising 7,540,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 2,700,000-4,000,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 10,000,000 individuals. Based on ringing data, it appears that some birds from Iceland, Faroe and

Norway visit UK waters. Therefore, a biogeographic population with connectivity to UK waters is the sum of numbers in UK, Iceland, Faroe and Norway (2,486,000 pairs). When accounting for immature birds, this represents a total of almost 5,000,000 adults and about 3,000,000 immatures; i.e. a total of about 8,000,000 birds.



Figure 5.2. Breeding population origins of fulmars in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

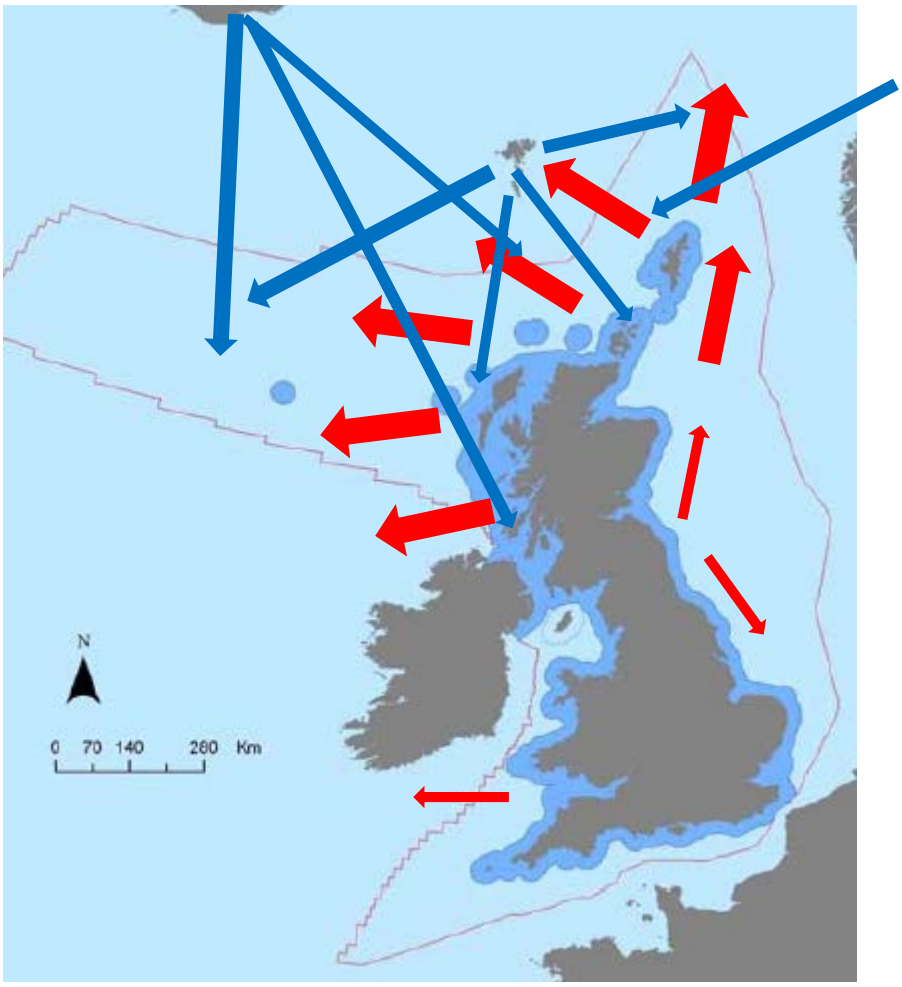


Figure 5.3. Main movements of fulmars from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure. There is only low to moderate confidence in these data, since none of these populations have been studied by tracking (apart from a very small number of birds in Orkney for which no data are yet in the public domain), and fulmar movements are not easy to determine because the species is so widely distributed and predominantly pelagic.

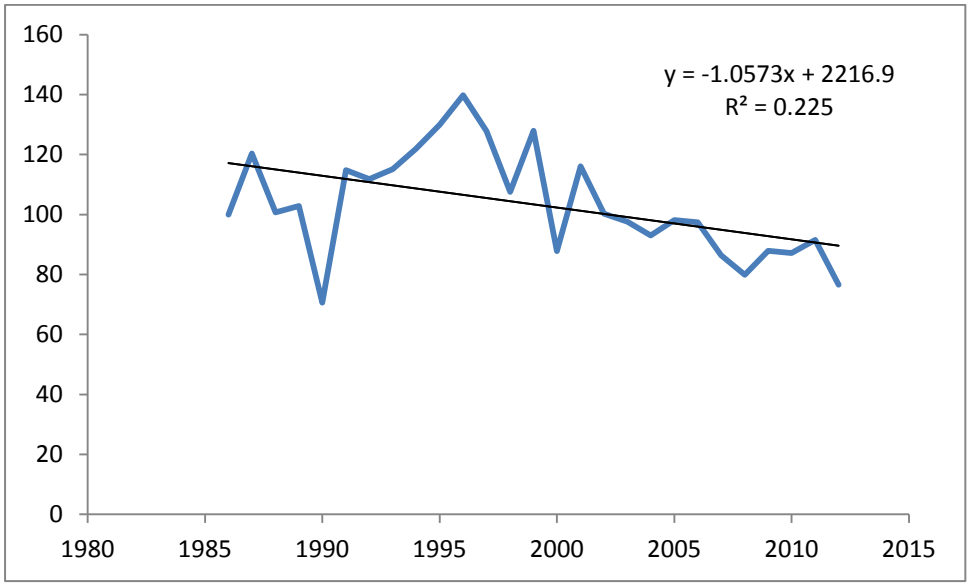


Figure 5.4. Trend in the fulmar breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

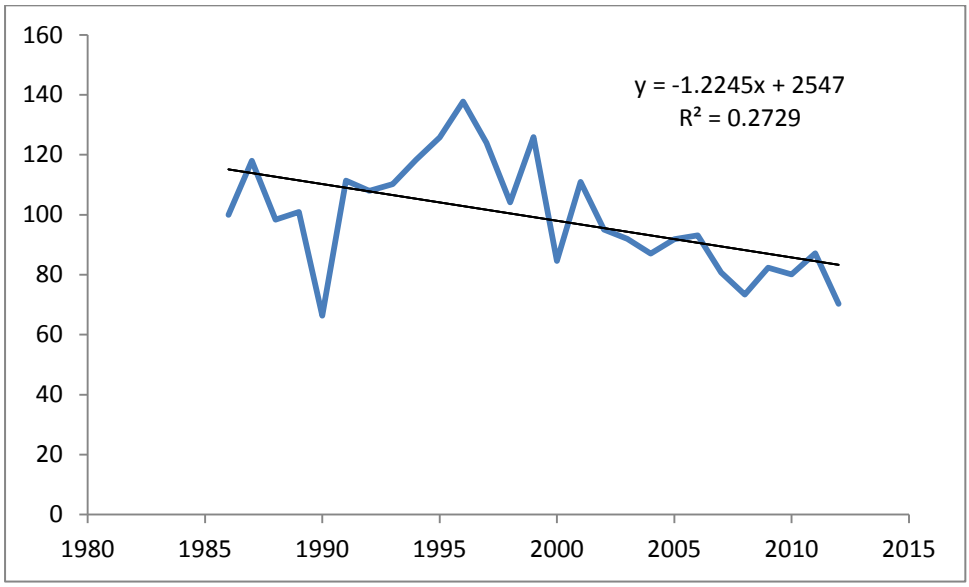


Figure 5.5. Trend in the fulmar breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

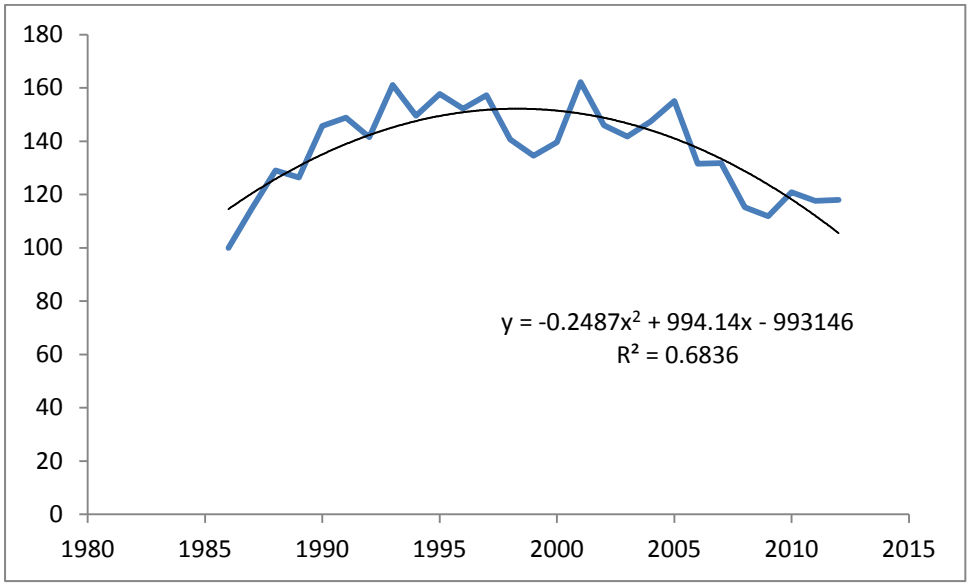


Figure 5.6. Trend in the fulmar breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

5.9 Proportion of UK population from UK breeding SPAs

The 25 SPAs with breeding fulmars as a feature together held 310,279 pairs at designation, estimated to represent ca. 57% of the British breeding population (Stroud et al. 2001). Given that the geographical distribution of SPAs (Figure 5.7) reflects the geographical distribution of the population as a whole, it is likely that this percentage remains a valid estimate for the current population. Breeding numbers have declined since 2000 (by about 10% between 2000 and 2012) in the UK, Scotland and Wales (Figures 5.4 to 5.6). However, that decline is likely to have affected SPA and non-SPA populations, so should not greatly alter the proportion within SPAs. It is likely that larger populations (which are predominantly the SPA populations) may have declined more, which would reduce the proportion within the SPA suite. Stroud et al. (2014) estimated that the SPA suite held about 49.7% of the GB population in the early 2000s.

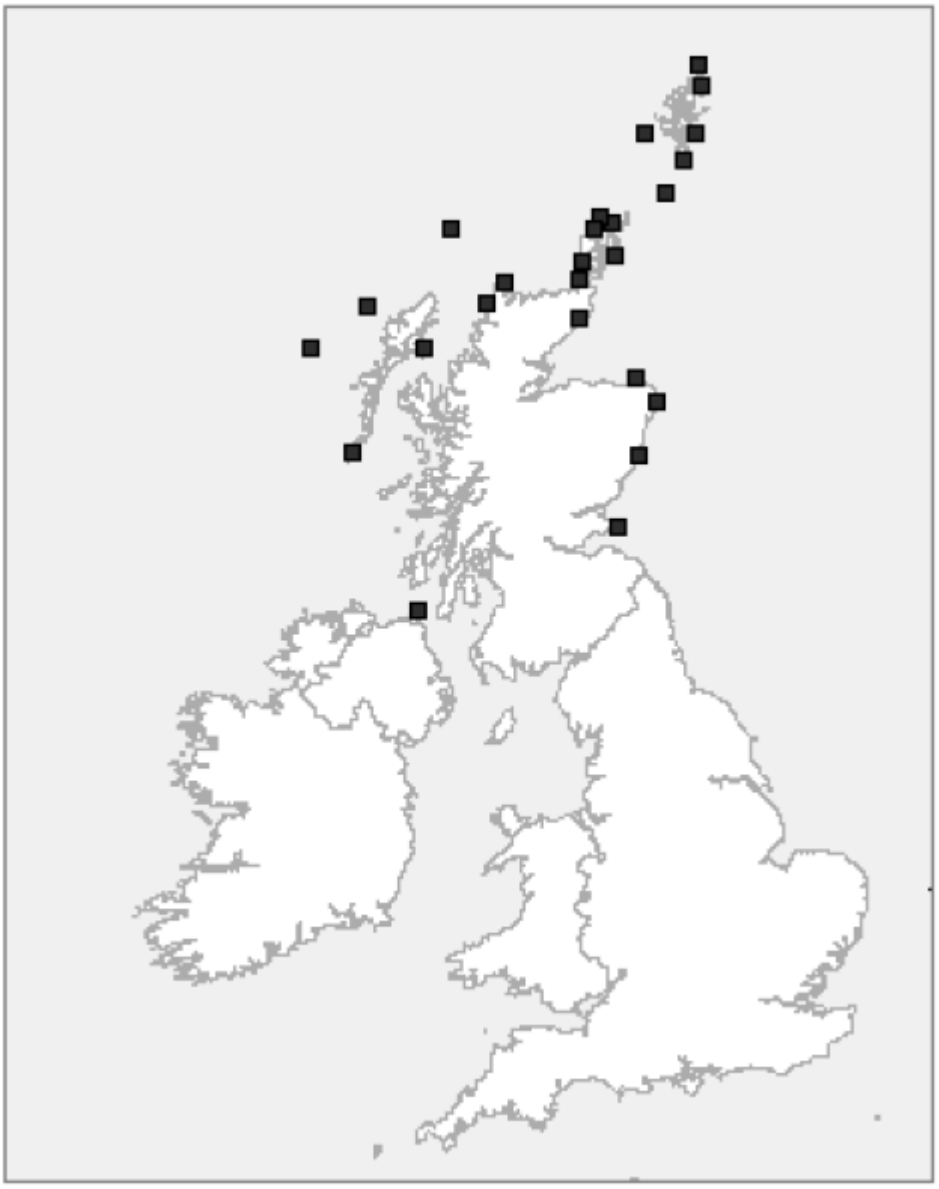


Figure 5.7. Locations of the 25 UK SPAs with fulmar as a breeding feature. These SPA populations are listed in Table 5.1. From Stroud et al. 1990.

Table 5.1. The UK SPA suite for breeding fulmars.

SPA	Location	Pairs	Year desig-nated	Site condition monitoring*	Recent counts (pairs)	Year	Reference
UK North Sea							
Hermaness, Saxavord & Valla F	Shetland	14,890	1994	Declined 2007	13,958 >6,723	1999 2011	SMP database SMP database
Fetlar	Shetland	9,800	1994	Maintained 2002	8,912	1999-2002	Stroud et al. 2014
Foula	Shetland	46,800	1995	Declined 2007	21,106 19,758	2000 2007	Seabird2000 SMP database
Noss	Shetland	5,870 (1993)	1996	Maintained 1998	4,999 5,169 6,144 5,248	1998 2002 2006 2011	Seabird2000 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012

Sumburgh Head	Shetland	2,542	1996	Maintained 2001	1,487 230 233	2001 2007 2009	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012
Fair Isle	Shetland	43,320	1994	Maintained 2000	29,649	2011	Lewis et al. 2012
West Westray	Orkney	1,400	1996	Declined 2007	4,270 677	2000 2007	Seabird2000 Lewis et al. 2012
Calf of Eday	Orkney	1,955	1998	Maintained 2002	1,842	2002	Lewis et al. 2012
Rousay	Orkney	1,240	2000	Recovering 2009	712 1,030	2000 2009	Seabird2000 Lewis et al. 2012
Hoy	Orkney	35,000	2000	Declined 2007	19,586	2007	Lewis et al. 2012
Copinsay	Orkney	1,615	1994	Recovering 2008	1,630	2008	Lewis et al. 2012
North Caithness Cliffs	N Scotland	14,700 Or 16,310 (Stroud et al. 2001)	1996	Maintained 2000	14,250	2000	Seabird2000
East Caithness Cliffs	N Scotland	15,000	1996	Maintained 1999	14,202	1999	Seabird2000
Buchan Ness to Collieston Coast	NE Scotland	1,765 (1986)	1998	Declined 2007	1,389 1,367	2007 2007	Lewis et al. 2012 Stroud et al. 2014
Troup, Pennan and Lion's Heads	NE Scotland	4,400 (1995)	1997	Declined 2007	2,900 1,795	2001 2007	Lewis et al. 2012 Lewis et al. 2012
Fowlsheugh	NE Scotland	1,170	1992	Maintained 1999	246 193	2006 2009	Lewis et al. 2012 Lewis et al. 2012
Forth Islands	E Scotland	798 (1985) or 1,600 (Stroud et al. 2001)	1990	Maintained 2004	1,364 676 832	2004 2005-2009 2010	Lewis et al. 2012 Stroud et al. 2014 Lewis et al. 2012
Flamborough and Filey Coast pSPA	E England	Not stated	Not yet		1,355 878	2000 2008	SCM database SCM database
Western waters & Channel							
Cape Wrath	NW Scotland	2,300	1996	Maintained 2000	2,115	2000	Seabird2000
Handa	NW Scotland	3,500 (1986)	1990	Declined 2008	4,323 3,550 2,119 1,915 1,870	1996 2000 2004 2008 2012	SMP database Seabird2000 Lewis et al. 2012 Lewis et al. 2012 SMP database
Flannan Isles	Western Isles	4,700 (1988)	1992	Recovering 2013	7,328	1998	Seabird2000
North Rona and Sula Sgeir	N Scotland	11,500	1985-1986	Declined 2012	North Rona only: 3,738 3,520 2,616 1,438	 1986 1998 2005 2012	 SMP database SMP database SMP database SMP database

Shiant Isles	Western Isles	6,820	1992	Maintained 1999	4,387	1999	Seabird2000
St Kilda	Western Isles	62,800	1992	Maintained 2000	66,055	1999	Seabird2000
Mingulay and Berneray	Western Isles	12,500 (1994)	1994	Maintained 1998	15,023 9,046	2003 2009	Lewis et al. 2012 Lewis et al. 2012
Rathlin Island	N Ireland	1,482 (1985)	1999		2,032 1,072 1,518	1999 2007 2011	SCM database SCM database SCM database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

5.10 BDMPS

Given that breeding fulmars from Scotland may make foraging trips while breeding to as far as the mid-Atlantic ridge, and non-breeding birds may disperse over thousands of kilometres, this pelagic species cannot readily be subdivided into local regional populations. It seems more appropriate to consider all UK waters as a single BDMPS for this species. However, if it is convenient to work on a smaller spatial scale, division into UK North Sea waters and UK Western waters plus Channel would be practical, based on the fact that there appears to be relatively low movement of birds between UK North Sea and UK western waters (Figure 5.8). The following interpretation is based on the review of literature presented in Sections 5.5, 5.6 and 5.7. In UK waters there are about 1,000,000 fulmars at sea on average during the winter. The vast majority of these occur in Scottish waters rather than further south. Approximately half of these occur in the BDMPS 'UK North Sea waters' and approximately half in 'UK Western waters plus Channel'. However, the contributions of SPA breeding populations differ strongly between these two BDMPS, with most birds from North Sea SPA populations in North Sea waters and most birds from western SPA populations in western waters. Details of apportioning used in computing these BDMPS are given in Appendix A Tables 8 and 9 for winter BDMPS, and Appendix A Tables 10 and 11 for migration season BDMPS. The numbers of birds from overseas populations contributing to these BDMPS is particularly uncertain. It is clearly a very low proportion as estimates of the numbers of fulmarsv at sea in UK waters would not allow for large numbers from overseas in addition to the better known numbers from UK populations. In the BDMPS calculations the proportion coming from Iceland, Norway and Faroe has been estimated at 1% of the adult population and 2% of the immature population (3% for Faroe) in winter in the UK North Sea and in UK western waters, and at twice these values for the migration seasons.

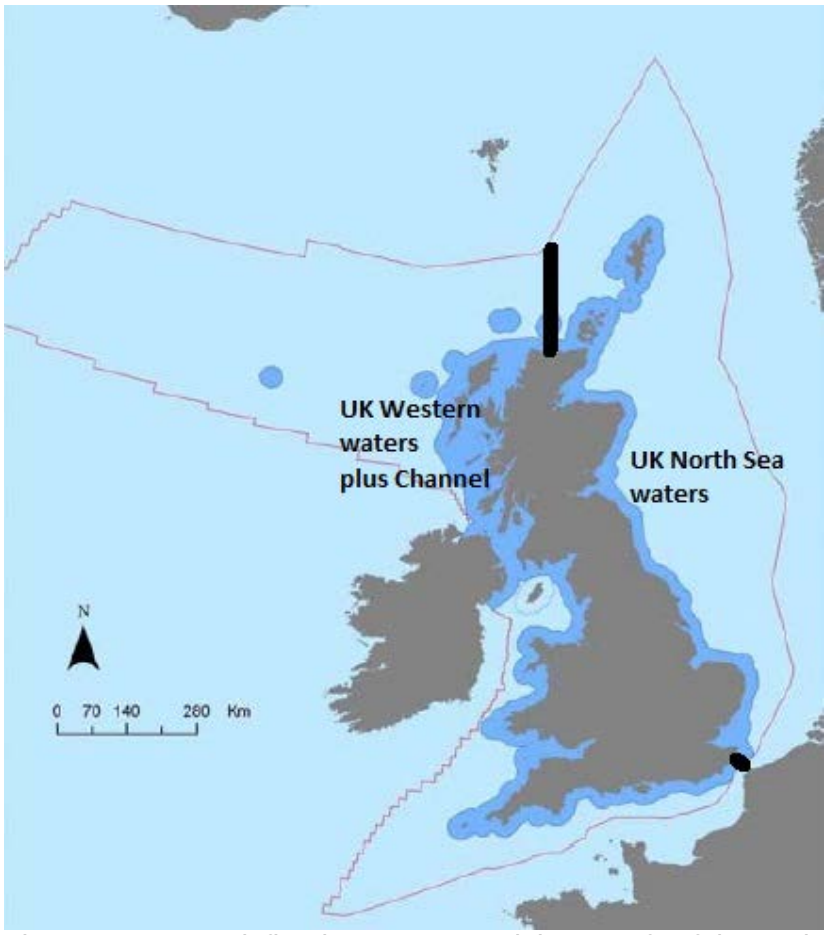


Figure 5.8. Two defined BDMPS spatial areas for fulmar; the two defined areas are 'UK North Sea' and 'UK Western waters plus Channel'.

5.11 Proportions of UK SPA birds in each BDMPS

Proportions of each BDMPS represented by adults from UK breeding SPA populations can be computed from the data in Appendix A Tables 8 to 11. For example, the UK North Sea winter BDMPS holds an estimated 96,413 birds from overseas populations and 472,323 birds from UK populations, a total of 568,736 birds. Of these, 184,608 are adults from SPA breeding populations, so these represent 32.5% of the UK North Sea winter BDMPS total. In UK western waters the winter BDMPS holds an estimated 96,413 birds from overseas populations and 459,954 birds from UK populations, a total of 556,367 birds. Of these, 162,063 are adults from SPA breeding populations, so these represent 29.1% of the UK western waters plus Channel winter BDMPS total.

5.12 Spatial distribution of UK breeding SPA birds across the BDMPS

The 25 UK SPAs with fulmar as a feature are almost all in Scotland (Figure 5.7), but this also reflects the broader breeding distribution of the species in the UK: Seabird 2000 (Mitchell et al. 2004) reported 485,852 pairs in Scotland, 9,755 in England, 3,474 in Wales and nearly 6,000 in Northern Ireland, so the Scottish population represents over 96% of the UK total. Within Scottish waters, the spread of fulmar SPAs is also distributed much as the overall breeding population, so that the at sea distribution of birds from SPA populations is likely to be very similar to that of birds from colonies that are not SPAs. Furthermore, about 50% of the fulmar population breeds on SPAs with fulmar as a designated feature, so the high proportion of the population in designated sites also makes it likely that the geographic spread of birds from SPAs matches closely that of the general population. The high mobility of this pelagic species also means that birds are likely to be well mixed at sea during migration seasons and in winter.

6. MANX SHEARWATER *Puffinus puffinus*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (August to early October and late March to May)
Overseas	242,000	11,206
UK	1,700,000	1,578,196
Total	2,000,000	1,589,402

Migration season BDMPS (August to early October, late March to May)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
UK North Sea waters	8,507	111	8,396
UK Western waters plus Channel	1,580,895	11,095	1,569,800

Colour coding is amber for western waters as numbers in colonies in the UK and overseas are moderately well known and have in most cases not been censused since Seabird 2000 (and there are some issues with estimated numbers at Skomer where recent census suggests surprisingly large increase in numbers), most birds in UK waters originate from UK colonies so the influence of uncertain numbers coming from overseas is relatively small, and movement patterns of this species appear to be consistent from year to year. Colour coding for the North Sea migrating BDMPS is red because numbers entering the North Sea are low, are not well documented, and seem to vary somewhat from year to year, possibly in response to variable weather conditions.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 12 and 13.

6.1 Breeding range and taxa

Manx shearwaters are monotypic, with a core breeding range in the British Isles, smaller populations in Faroe and Iceland, and very small colonies in eastern Canada, France, Azores, Madeira and Canaries. Biometric variation appears to be of no value in assessing origins of individuals. Manx shearwaters are trans-equatorial migrants, wintering off the coast of Brazil (Brooke 1990).

6.2 Non-breeding component of the population

Manx shearwaters start to breed when 5 years old (BTO Birdfacts). Adult survival rate is 0.905 (BTO Birdfacts), juvenile survival 0.25 up to 5 years old (BTO Birdfacts) and mean productivity is 0.591 chicks per pair (JNCC database, n=56 measurements). To obtain a stable population, survival of immatures was adjusted to 0.6 for juveniles, 0.8 for 1-year olds,

0.85 for 2-year olds, 0.88 for 3-year olds, and 0.9 for 4-year olds. The model population comprised 54% adults, 16% juveniles and 30% older immatures. There are 0.84 immatures per adult.

6.3 Phenology

Some chicks may still be emerging and fledging from burrows on Rum in mid-October after adults have departed. However, most adults leave the breeding colonies by late September or early October (Pennington et al. 2004; Forrester et al. 2007). The literature indicates that autumn migration starts in July (Cramp et al. 1977-94; Pennington et al. 2004), August (Forrester et al. 2007), or mid-August (Wernham et al. 2002). Peak autumn migration occurs in August in Shetland (Pennington et al. 2004), August-October throughout the range from Europe to South America (Cramp et al. 1977-94) or in September in the UK (Wernham et al. 2002; Forrester et al. 2007). Argyll Bird Reports indicate very large movements of Manx shearwaters through Argyll waters in August each year (flocks of tens of thousands of birds) but only small numbers in September and very few in October. It is unclear whether this means that migration mainly occurs in August (whenm chicks are still in burrows) or whether these very large movements are foraging by breeding adults rather than migration movements. If the latter, this would imply that migration occurs rather directly into the Atlantic so is not evident from coastal Argyll for example. Numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) suggest that autumn migration occurred mainly in September with a little in early October (Figure 6.1). Autumn migration is completed by late September (Pennington et al. 2004; Forrester et al. 2007) or early October (Cramp et al. 1977-94; Wernham et al. 2002; Brown and Grice 2005).

Spring migration starts from South America in mid-January (Cramp et al. 1977-94), and in UK waters in February-March (Wernham et al. 2002) or early March (Pennington et al. 2004; Forrester et al. 2007). Peak spring migration occurs in February-March through in the entire range of the species (Cramp et al. 1977-94), but in mid-March in English waters (Brown and Grice 2005), in late March according to Forrester et al. (2007), April according to Wernham et al. (2002), or May in Shetland (Pennington et al. 2004). Numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) suggest that spring migration occurred in April-May (Figure 6.1). Spring migration is completed by April (Cramp et al. 1977-94), late April (Wernham et al. 2002), May (Forrester et al. 2007) or as late as June in Shetland (Pennington et al. 2004).

The first spring records of Manx shearwater in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly from late March to late April, and the last records were predominantly in late September or October, while peak autumn migration was reported in July, August or September in most years, and peak spring migration was reported in May in most years. Breeding colonies are first re-occupied in March or April, with modal arrival at colonies in late March or April (Pennington et al. 2004; Forrester et al. 2007).

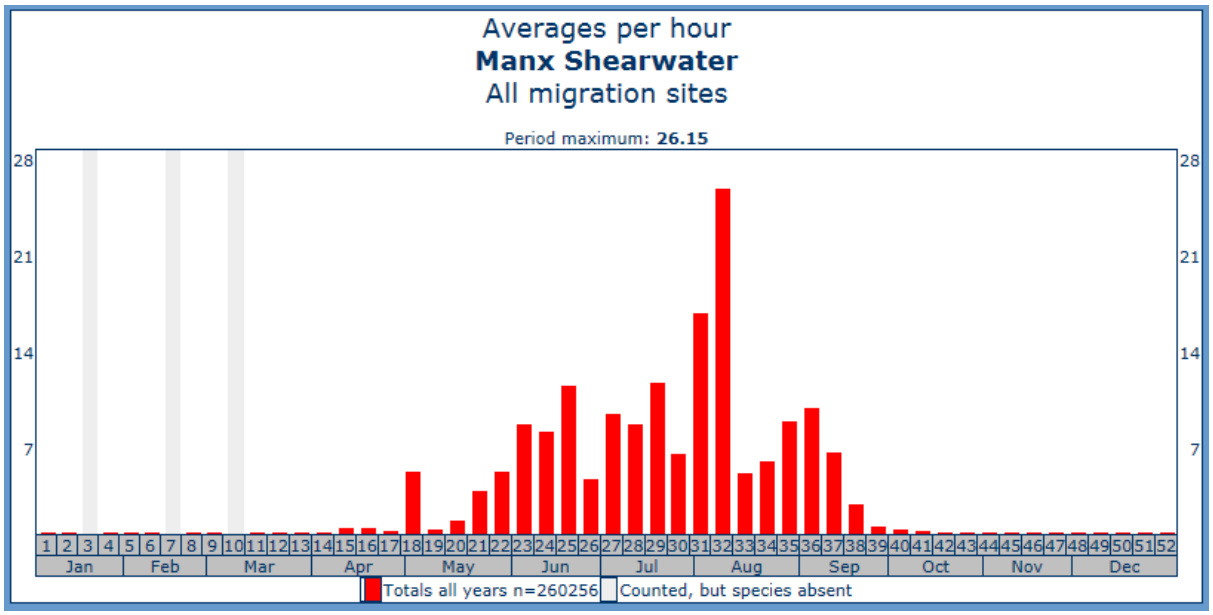


Figure 6.1. Average numbers of Manx shearwaters counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-September, non-breeding season birds absent from UK waters. However, from the data reviewed above, a more appropriate definition would be breeding season April-August, non-breeding season September-March.

6.4 Defined seasons:

- UK Breeding season — April-August
- Post-breeding migration in UK waters — August-early October (**migration BDMPS**)
- non-breeding season — September-March
- Return migration through UK waters — late March-May (**migration BDMPS**)
- Migration-free breeding season — June-July
- Migration-free winter season — November-February

Apart from the breeding season, one seasonal BDMPS periods is considered to be appropriate for Manx shearwater:

Migration seasons BDMPS (August-early October and late March-May).

6.5 Movements of birds from the UK population

Birds from UK colonies depart in August to October, apparently predominantly in September, and most reach South America by October (Brooke 1990; Wernham et al. 2002). Indeed, there are recoveries of chicks that have reached South America within two to three weeks of the date of ringing at the nest (Brooke 1990). Birds from Rum are thought to migrate predominantly past the west of Ireland rather than through the Irish Sea (supported by the records of Argyll Bird Club that very large numbers of Manx shearwaters feed in Argyll waters in August but rather few tende to be seen in Argyll waters in September and hardly any in October), and then past France and Spain and probably past west Africa before crossing to South America (Wernham et al. 2002). Spring migration appears to follow a more westerly route (Brooke 1990). Large numbers are seen off North Carolina in February-March (Wernham et al. 2002). There is some evidence to suggest that the use of waters off the United States is a feature that has developed since the 1950s, as the species was largely

unknown there in earlier decades (Brooke 1990), suggesting some flexibility in migration route, perhaps in response to changing environmental conditions. Some immature birds, predominantly birds that are only one year old, remain in wintering areas or off the southeastern United States rather than returning to British waters (Wernham et al. 2002). Movements of adults through the South and North Atlantic have been tracked by geolocator deployment, but although these provide clear evidence of the large scale pattern they give only very little indication of directions of migration movements through UK waters (Guilford et al. 2009).

6.6 Movements of birds from overseas into UK waters

Of the 1,036 birds ringed in the Faroes (357 as chicks) none have been recovered in Britain (Hammer et al. 2013). According to Wernham et al. (2002), there is no evidence from ringing, or from any other sources, to suggest that birds from colonies in other countries apart from Ireland pass through British waters during migration, although a small number of birds reared in French colonies have recruited into colonies in the UK. However, it seems highly likely that most birds from the Faroes pass through the NW area of UK territorial waters on migration, and some from Iceland may do so. Most birds from Irish colonies probably migrate directly between the open Atlantic Ocean and Irish waters rather than moving through UK waters. There are probably about 400,000 pairs in UK colonies (numbers being somewhat uncertain due to variations in recent counts at the largest colonies), 32,600 pairs in Ireland, 25,000 pairs in Faroes, and 8,500 pairs in Iceland (Mitchell et al. 2004). These data would suggest that all, or almost all, of the Manx shearwaters occurring in British waters during migration are from British colonies. Although there are occasional records of Manx shearwaters in British waters as late as November or December, these are highly unusual, and no birds are thought to overwinter successfully in British waters.

6.7 Numbers in UK waters

Very high densities occur in summer (May-August) in Irish and Celtic Sea, whereas the species is scarce in the Channel and in the North Sea (Brown and Grice 2005). Forrester et al. (2007) suggest that passage of Manx shearwaters through Scottish waters is ‘minimal’ apart from the arrival and departure of birds to and from the large colonies on Rum and St Kilda. This is supported by the very small numbers of migrant Manx shearwaters seen at Shetland or Orkney or along the east coast of the Scottish mainland, where the species has no significant breeding colonies (Annual Bird Reports and Pennington et al. 2004).

6.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the entire species’ population, comprising 265,100 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 340,000-410,000 pairs. Since then, the estimated large increase in numbers at Skomer suggests that a more appropriate total may be at least 400,000 pairs for the UK, possibly slightly more. Kober et al. (2010) presented an estimated biogeographic population of 1,130,000 individuals. Since populations in the UK, Iceland, Faroe and Ireland have possible connectivity with UK waters, the appropriate biogeographic breeding population with connectivity is a total population of ca. 2,000,000 birds. However, given that there is no evidence that Manx shearwaters from Ireland, Iceland and Faroe pass though UK coastal waters, and these birds are considerably outnumbered by the UK population, it would be a reasonable first approximation to consider all Manx shearwaters occurring in UK waters to be birds from the UK population, comprising ca. 400,000 pairs (800,000 adults) and an associated 672,000 immatures. Some of the younger immatures spend the entire year in the wintering area (off South America) so that perhaps 1,580,000 birds from UK colonies plus about 11,200 from overseas colonies may be in UK waters during the migration periods. Numbers breeding at Rum are not known with confidence as that (very large) colony is very difficult to census, and trends in breeding numbers are

unclear for Rum, and for other major colonies (Table 6.1). Numbers at Skomer are also somewhat uncertain due to the estimated large increase in numbers there when a new census methodology was adopted (Perrins et al. 2012).



Figure 6.2. Breeding population origins of Manx shearwaters in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

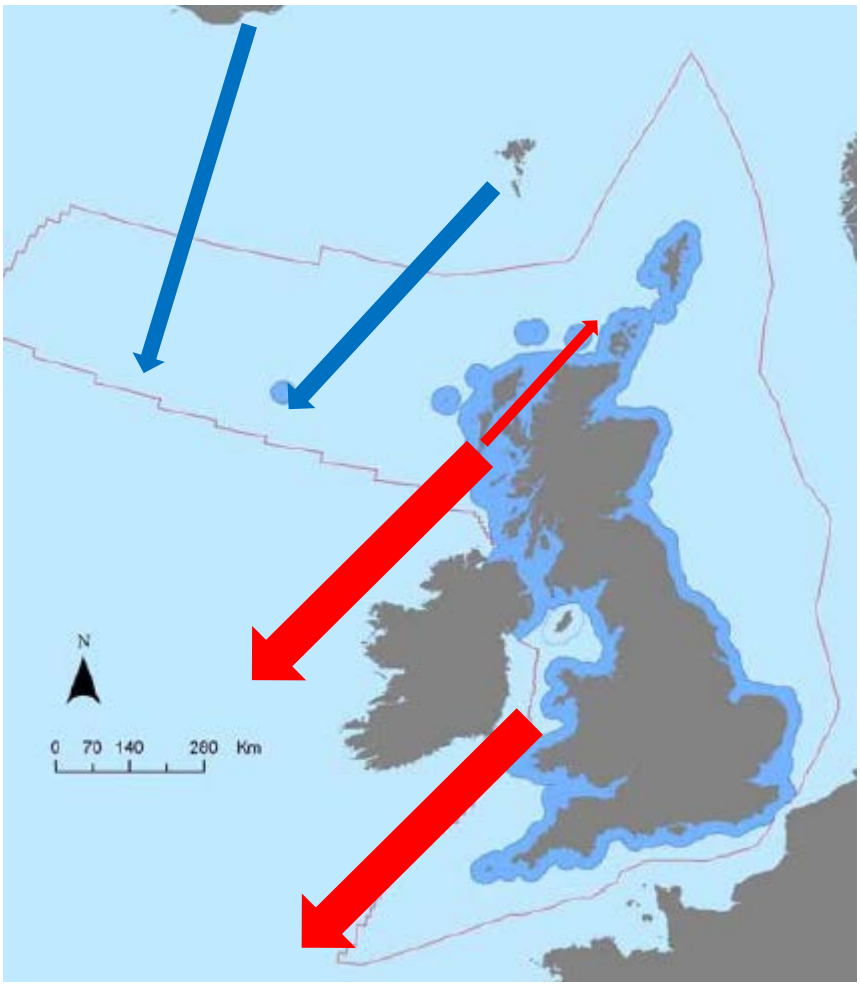


Figure 6.3. Main movements of Manx shearwaters from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure, except that in spring it is thought that birds tend to arrive from further west, crossing the North Atlantic from the Grand Banks area.

6.9 Proportion of UK population from UK breeding SPAs

The 4 SPAs with breeding Manx shearwaters as a feature together held 219,898 pairs at designation, estimated to represent ca. 100% of the British breeding population (Stroud et al. 2001). These SPA populations continue to represent almost the entire UK population. Stroud et al. (2014) estimated that GB SPAs held 96.2% of the GB population in the early 2000s. However, a recent census of Skomer found an estimated 316,070 breeding pairs on that island, more than twice the expected number (Perrins et al. 2012), suggesting that the total for the Skokholm, Skomer and Middleholm SPA is likely to have reached about 350,000 pairs, considerably increasing the total estimated UK (and world) population size. If this recent census is confirmed to be accurate (it used a new census method but is thought by Perrins et al. 2012 to be appropriate) this implies that the UK population of Manx shearwaters is at least 400,000 pairs, and possibly higher. The UK SPA suite for breeding Manx shearwaters still certainly holds very near to 100% of the UK breeding population.

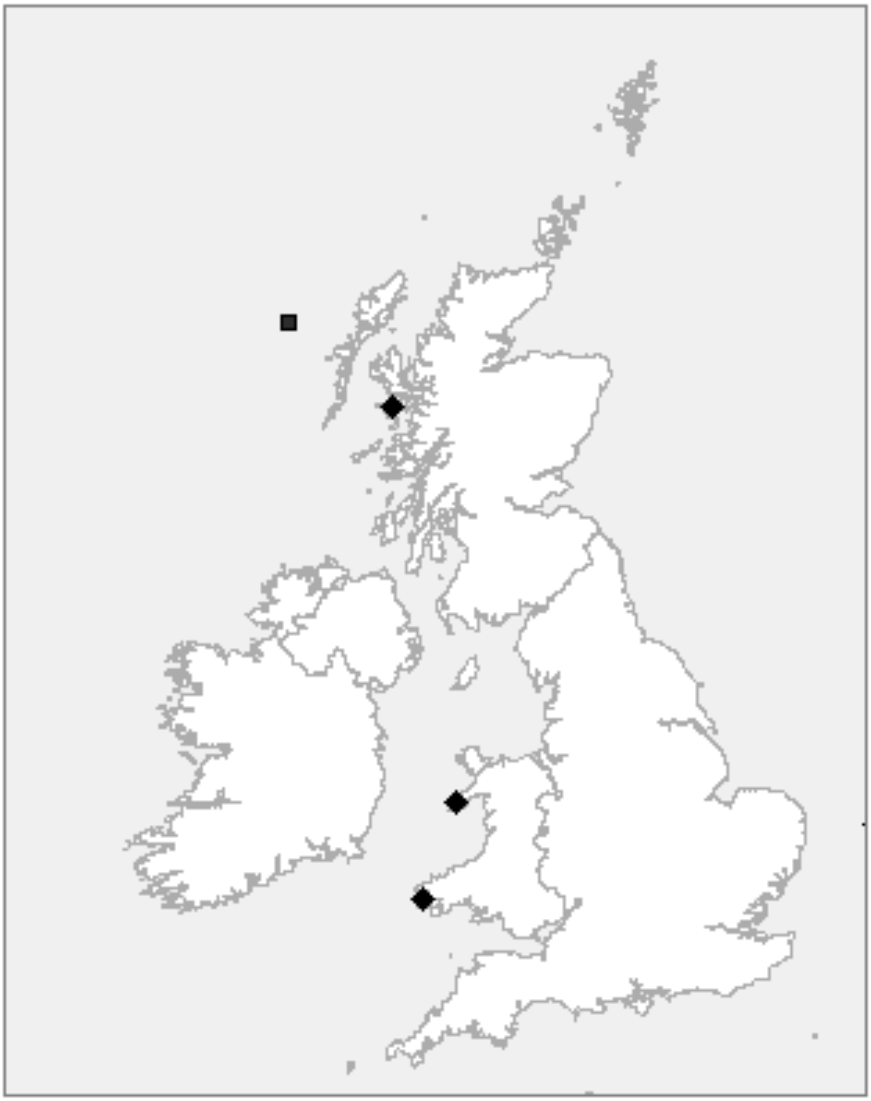


Figure 6.4. Locations of the 4 UK SPAs with Manx shearwater as a breeding feature. These SPA populations are listed in Table 6.1. From Stroud et al. 1990.

Table 6.1. The UK SPA suite for breeding Manx shearwaters.

SPA	Location	Pairs	Year desig-nated	Site condition monitoring*	Recent counts (pairs)	Year	Reference
Western waters & Channel							
St Kilda	Western Isles NW	<5,000 or 1,000 (Stroud et al. 2001)	1992	Maintained 2000	4,802	1999	Seabird2000
Rum	Inner Hebrides NW	61,000 (1995)	1982	Maintained 2003	120,000	2001	Seabird2000
Aberdaron Coast & Bardsey Island	Wales SW	6,930 (1996)	1992		16,183	2001	SCM database
Skomer, Skokholm & Middleholm	Wales SW	150,968 (1998)	1982		350,000	2011	Perrins et al. 2012 and in litt.

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

6.10 BDMPS

The following interpretation is based on the review of literature summarised in sections 6.5, 6.6 and 6.7. Although Manx shearwater is a highly pelagic species, the migration of this species out of, and back into UK waters appears to take place fairly quickly and directly. The BDMPS 'UK Western waters plus Channel' holds the large colonies in Wales, on Rum and St Kilda and a few small colonies. The BDMPS 'UK North Sea waters' holds no large colonies and no SPA breeding populations of the species and has very few migrant Manx shearwaters passing through. All these areas hold no birds in winter, so the BDMPS of concern is that for migration seasons. During migration, there will be about 1.6 million passing through the 'UK Western waters plus Channel' area. Numbers passing through the 'UK North Sea waters' are low, possibly around 8,000-9,000 birds but varying from year to year and often much less than this. Details of apportioning are given in Appendix A Tables 12 and 13. It is estimated that only about 1% of immatures and no adults from UK SPA colonies, 1% of adults and immatures from UK non-SPA colonies, and 0.1% of immatures and no adults from Iceland, Faroe and Ireland migrate through UK North Sea waters, while 100% of adults and 70% of immatures from UK SPA colonies migrate through UK western waters, together with 80% of adults and 60% of immatures from UK non-SPA colonies (numbers in these colonies being trivial by comparison to numbers in SPA colonies), and 1% of adults and 3% of immatures from Iceland and Faroe, 5% of adults and 10% of immatures from Ireland. Numbers of immatures in UK waters (and so components of these two BDMPS) do not sum to 100% because many of the youngest immatures remain in South American waters until at least their second year.

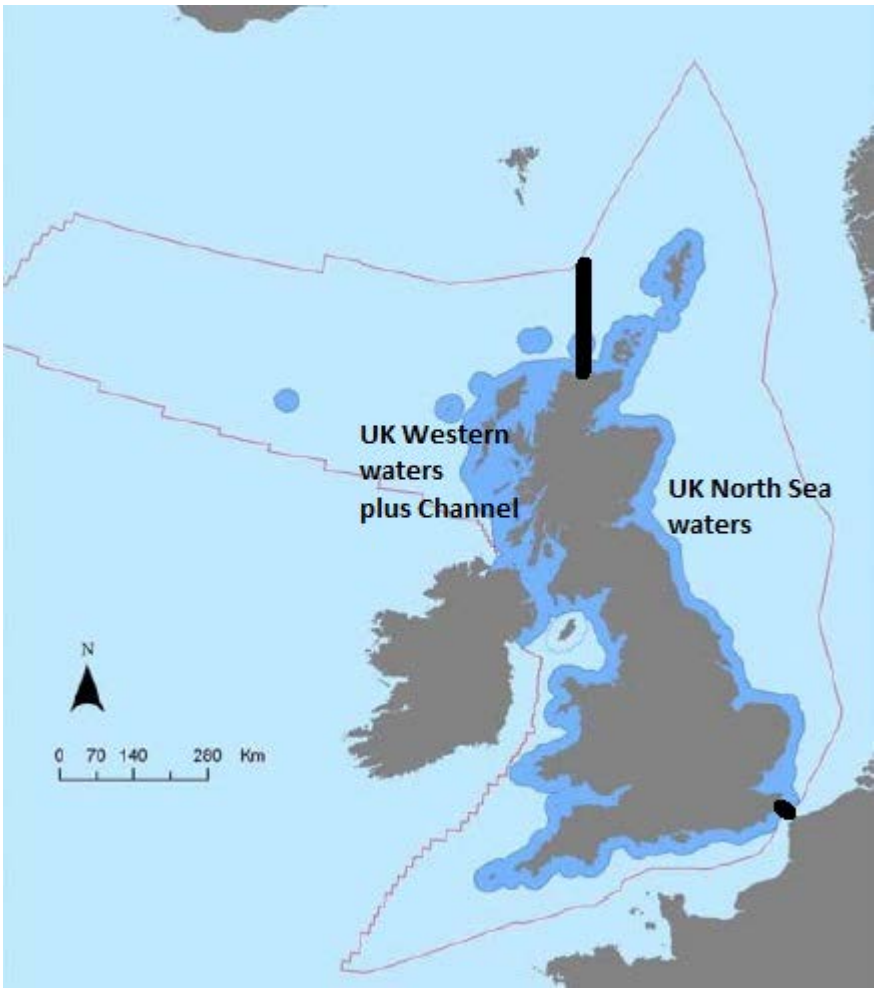


Figure 6.5. Two defined BDMPS spatial areas for Manx shearwater. The two areas are: 'UK North Sea waters' and 'UK Western waters plus Channel'.

6.11 Proportions of birds from BDMPS in reference regions

The vast majority of all birds found in these two BDMPS are associated with UK colonies. A very small number of birds migrate through from or towards colonies in Ireland, Iceland or Faroe, but those numbers are trivial based on the limited evidence. Almost all of the birds are from UK colonies and almost all birds in UK colonies are in SPA populations. The proportion of the BDMPS that comprises adults from SPA populations can be computed from Appendix A Tables 12 and 13. In the UK North Sea BDMPS of 8,507 birds, none are thought to be adults from SPA populations since the small numbers passing through the North Sea are most likely to be immatures rather than breeders, or birds from Faroe and Iceland. In the UK Western Waters plus Channel BDMPS of 1,580,895 birds, 981,970 are estimated to be adults from SPA breeding populations, or 62% of the total (most of the rest being immatures that originated from these SPA colonies).

6.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Since virtually 100% of Manx shearwaters in UK colonies are in SPAs with Manx shearwater as a feature, the spatial distribution of SPA birds is virtually identical to that of the population as a whole.

7. NORTHERN GANNET *Morus bassanus*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in autumn (September-November) (adults and immatures)	Numbers in UK waters in spring (December-March) (adults and immatures)
Overseas	260,000	108,522	87,606
UK	923,000	893,730	822,667
Total	1,180,000	1,002,252	910,273

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK breeding population (adults plus immatures)
'Autumn migration' BDMPS (September to November)			
UK North Sea and Channel	456,298	45,173	411,125
UK Western waters	545,954	63,349	482,605
'Spring migration' BDMPS (December to March)			
UK North Sea and Channel	248,385	21,903	226,482
UK Western waters	661,888	65,703	596,185

Colour coding is green for numbers of birds in UK waters since the numbers are based on rather accurately known breeding numbers in UK colonies, and match quite well with estimates of numbers at sea from ESAS and general literature (such as Forrester et al. 2007). Movements of UK gannets are well known from ringing and are less subject to recovery bias than for more pelagic seabird species. Numbers visiting UK waters from overseas populations are certainly much smaller than numbers from UK colonies, but are less certain. There have been studies tracking migrating gannets (deploying geolocators) from colonies in Norway and Iceland which indicate movement of adults from those populations into and through UK waters, and ringing data also show connectivity, but the proportion of birds from those populations visiting UK waters is rather uncertain. However, given that numbers from overseas populations coming into UK waters are undoubtedly small relative to numbers from UK colonies, overall total numbers are coded green because those are mainly determined by numbers from UK colonies. Due to extensive tracking studies of breeding adults from many different colonies in different countries, confidence in the

movement patterns of gannets is high. However, details of the movements of immature birds are less well known, although the general pattern appears to be similar to that of adults but with immatures moving further south on average, and migrating later in spring, with youngest immatures remaining in wintering areas. There is some uncertainty about numbers at sea because much survey work that was boat-based involved data that appear to be biased by the stong attraction of gannets towards boats.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 14 to 17.

7.1 Breeding range and taxa

Gannet is a monotypic species with core breeding range within the British Isles, but colonies also in Norway, Russia, Faroe, Iceland, eastern Canada, Germany and France. Biometrics do not seem to vary significantly among populations.

7.2 Non-breeding component of the population

Gannets start to breed when 5 years old (BTO Birdfacts; WWT 2012). Adult survival rate is 0.92 (BTO Birdfacts; WWT 2012), juvenile survival 0.42 (BTO Birdfacts) and mean productivity is 0.684 chicks per pair (JNCC database, n=97 measurements). Survival of immatures was retained at 0.42 for juveniles, 0.83 for 1-year olds, 0.89 for 2-year olds, and 0.92 for older age classes. The model population comprised 55% adults, 19% juveniles and 26% older immatures. There are 0.81 immatures per adult.

7.3 Phenology

Breeding colonies are not completely deserted until mid-November, but modal departure occurs in late September (Pennington et al. 2004; Forrester et al. 2007). However, autumn dispersal/migration starts in August (Cramp et al. 1977-94; Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Peak autumn migration occurs in September in Shetland (Pennington et al. 2004) and in English waters (Brown and Grice 2005), late September in Scotland (Forrester et al. 2007), September-October in the UK (Wernham et al. 2002), September-November throughout Europe (Cramp et al. 1977-94), and October in Belgium (Vanermen et al. 2013). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in mid-September although seasonal pattern was not very pronounced in that data set (Figure 7.1). Autumn migration is completed by November (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007) or December if considering southern areas of Europe as well (Cramp et al. 1977-94).

Spring migration starts in December- January (Wernham et al. 2002; Pennington et al. 2004) early January (Forrester et al. 2007) or January (Cramp et al. 1977-94). Peak spring migration occurs in February-March (Pennington et al. 2004), February-April in Belgium (Vanermen et al. 2013), early March (Forrester et al. 2007), March (Wernham et al. 2002) or March-April (Cramp et al. 1977-94). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late-January and February (Figure 7.1). Spring migration is completed by late March (Pennington et al. 2004; Forrester et al. 2007) or early May (Wernham et al. 2002) or May (Cramp et al. 1977-94).

The first spring records of gannet in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly in early January and the last records were predominantly in late December, as some gannets overwinter, while peak autumn migration was reported in August to October in most years, and peak spring migration was reported in March or March-April in most years, but sometimes in January or February. Breeding sites are re-

occupied from early January, with modal re-occupation in mid-February to mid-March (Pennington et al. 2004; Forrester et al. 2007).

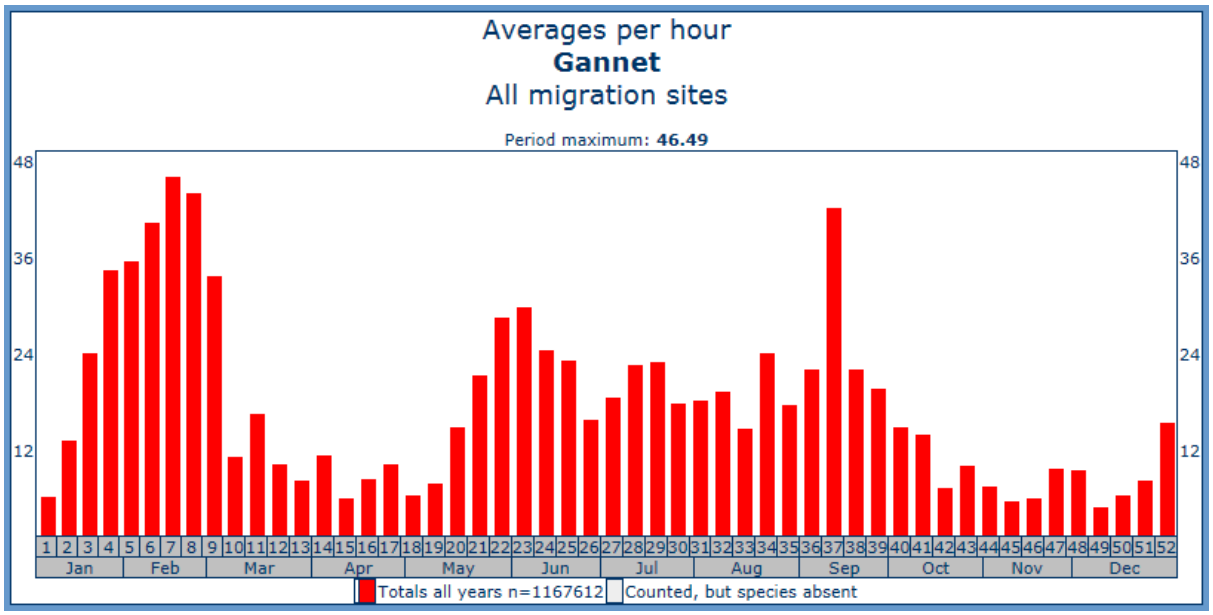


Figure 7.1. Average numbers of gannets counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-September, non-breeding season October-April. However, from the data reviewed above, a more appropriate definition would be breeding season March-September, non-breeding season October-February.

7.4 Defined seasons:

- UK Breeding season — March-September
  - Migration-free breeding season — April-August
- Non-breeding season — October-February
  - Post-breeding migration in UK waters — September-November (**autumn BDMPS**[2])
  - Migration-free winter season — None
  - Return migration through UK waters — December-March (**spring BDMPS**)

Apart from the breeding season, two seasonal BDMPS periods are considered to be appropriate for northern gannet:

'Autumn' (post-breeding) migration season BDMPS (September-November); and

'Spring' (pre-breeding) migration season BDMPS (December-March).

7.5 Movements of birds from the UK population

Gannets leave colonies mainly in August-October. Chicks fledge with large fat stores and begin migration by swimming, independent from their parents (Wernham et al. 2002) until their fat load is reduced. Fledglings generally move south quite rapidly; for example, birds ringed on the sea below the colony on Noss moved an average of 60 km per day during their

[2] Seasons for which BDMPS have been generated are annotated (BDMPS).

first 10-16 days (Wanless and Okill 1994) so clearly do not remain flightless for long. Adults from colonies in the UK do not necessarily move directly southwards in autumn, but may move to areas with abundant food for some time in late summer before heading towards their wintering area. On the basis of ring recovery data and observations of gannets on migration and in winter, Nelson (1978, 2002) suggested that most gannets breeding at the Bass Rock probably spend the winter in the North Sea or no further south than the Channel. Geolocators were fitted to experienced breeding gannets on the Bass Rock in 2002 and 2003 (Kubetzki et al. 2009). Birds attended the colony until between 24 September and 16 October (median 5 October). Although gannets fly at an average speed of about 58 km per hour (Garthe et al. 2007), migration took up to four weeks to complete, as birds spent considerable amounts of time sitting on the water or foraging locally rather than travelling consistently towards their goal, so net movement was often only 200 to 400 km per day. Of the 22 birds tracked until at least December, 18% wintered in the North Sea and the English Channel, 27% in the Bay of Biscay and the Celtic Sea, 9% in the Mediterranean Sea and 45% off West Africa. Birds wintering off West Africa migrated to their wintering areas mostly within 3 to 5 weeks, usually starting between early and late October. Most of these birds stayed off West Africa for a period of about 3 months, where they remained in a relatively restricted area. Individual winter home ranges as measured by the 75% kernel density contours varied between 8100 and 308 500 km^2 (mean = 134 000 km^2). Return migration was initiated between the end of January and mid-February, and took about as long as autumn migration. Kubetzki et al. (2009) inferred that the migration habits of gannets may be changing in response to human impacts on marine ecosystems, as the proportion of Bass Rock breeding adults that wintered within the North Sea was much smaller than appears to have been the case in earlier decades, whereas increased proportions were wintering off west Africa, where adult plumaged gannets had previously been relatively scarce. This trend was even more evident when loggers were deployed on Bass Rock gannets in 2008; none of the birds overwintered as far north as the North Sea that year (Garthe et al. 2012). These results are in strong contrast to the previously established view that adult gannets from the Bass Rock predominantly winter in the North Sea and only extremely exceptionally travel as far as Africa. Kubetzki et al. (2009) suggest that gannet migration behaviour may have changed in recent years, in response to changes in fish stocks and fisheries. In particular, amounts of fish discarded in the North Sea have been drastically reduced in recent years, whereas large fisheries have developed on the west African continental shelf and large quantities of discards are generated in that region (Meraz Hernando 2011). Almost all gannets (over 88%) seen on the west African shelf occur behind fishing vessels (Camphuysen and van der Meer 2005). In support of this suggested change in gannet winter distribution, Garthe (unpublished) analysed the ESAS database and found that the numbers of adult-plumaged gannets present in the North Sea in winter have declined since the 1980s despite very large increases in the gannet population. None of the birds carrying loggers wintered over deep water; all were on the continental shelf sea, wintering in areas where there are large fisheries as well as large stocks of pelagic fish (Meraz Hernando 2011). For birds where the logger data indicated migration routes used by breeding adults from the Bass Rock, twelve individuals migrated southwards through the English Channel, and eight left the North Sea around the north coast of Scotland and flew southwards west of the British Isles. On spring northward migration, only three birds moved back into the North Sea through the English Channel, while six moved into the North Sea around the north of Scotland (some loggers failed to record spring migration route because battery power was depleted). Birds that left in autumn through the Channel did not consistently return by the same route but in several cases moved north by a westerly route. A further deployment of loggers on Bass breeding adults in summer 2008 showed similar results (Garthe et al. 2010). On southward migration, 14 left the North Sea through the English Channel, and seven around the north of Scotland (apparently none of these birds flew overland from the North Sea to the Irish Sea or Atlantic). On northward migration in early spring, five entered the North Sea through the English Channel, and 16 flew up the west coast of Ireland and into the North Sea around the north of Scotland. Wernham et al. (2002) concluded that

distributions of gannet ring recoveries from different areas of Britain and Ireland, and recoveries from other European countries, show that gannets from all east Atlantic colonies intermingle in winter, distributed over a large area from the North Sea to west Africa. There is no clear evidence from ringing data that gannets from colonies in Britain and Ireland show differences among colonies in their wintering areas (Thomson 1974; Veron 1988; Wernham et al. 2002; Veron and Lawlor 2009). However, ringing effort has been high at the Bass Rock, moderate at Ailsa Craig, Hermaness, Grassholm and Great Saltee, and low or non-existent at other colonies. In particular, very little gannet ringing has been done at St Kilda, Sule Stack or Sula Sgeir, long-established and large colonies that represent a high proportion of the population and that are all located in the NW of the British Isles.

7.6 Movements of birds from overseas into UK waters

Ring recoveries from Faroese gannets suggest that those birds also share much the same winter distribution, but half of those birds were recovered as juveniles and wintering areas of adults were thought to be further north than most of these recoveries (Hammer et al. 2013). Deployment of geolocation loggers on breeding adult gannets from a variety of colonies showed evidence of different wintering areas used by birds from particular populations (Fort et al. 2012), although birds from all studied colonies were in UK waters in October. Their analysis showed that maximum distance between the colony and wintering area was similar across colonies despite their wide latitudinal range, strongly suggesting oriented chain migration (a pattern in which populations move uniformly southward). About 50% of the winter position fixes of birds from two Norwegian colonies were in UK waters (in the North Sea, west of Scotland, Channel, and Celtic Sea; see also Pettex et al. 2010). About 15% of the winter position fixes of birds from the Bass Rock were in UK waters (in the southern North Sea, Channel, and Celtic Sea). About 15% of the winter position fixes of birds from Rouzic (France) were in UK waters (almost all in the Celtic Sea). Less than 5% of the winter position fixes of birds from Grassholm were in UK waters. More recently, 12 loggers deployed on gannets at a colony in Iceland in summer 2010 were recovered in summer 2011 and preliminary analysis of these loggers indicates that the Icelandic gannets wintered from west Africa to west of Scotland (Garthe, Furness, Montevecchi and Halgrimsson unpublished data). During autumn migration, some of these birds passed through the North Sea and English Channel (5 out of 12) whereas in spring all returned northwards past the west of Ireland. Ringing studies indicate that immature gannets tend to winter further south than adults from the same population (Wernham et al. 2002). Wintering areas used by gannets breeding at colonies in Shetland and off NW Scotland have not been determined; no birds from those colonies have been equipped with geolocators and very few have been ringed. However, it seems likely that they will show patterns intermediate between colonies to the north (Norway and Iceland) and colonies to the south (Bass Rock, Grassholm, Rouzic). These data would suggest that a relatively small proportion of adult gannets from UK colonies overwinter in UK waters (and an even smaller proportion of immatures), whereas a relatively high proportion of adult gannets (but small proportion of immatures) from Norwegian and Icelandic colonies overwinter in UK waters. There are around 220,000 pairs in UK colonies, 36,000 pairs in Ireland, 5,950 pairs in the Channel Islands, 17,000 pairs in France, 28,500 pairs in Iceland, and 4,500 pairs in Norway (Wanless et al. 2005), 2,500 pairs in the Faroes (Hammer et al. 2013), about 632 pairs in Germany (Helgoland) (J. Dierschke in litt to JNCC July 2013) and a handful of pairs in Russia (Wanless et al. 2005). The fact that the UK population is by far the largest of these suggests that most gannets overwintering in southern UK waters are probably from UK colonies, whereas in the North Sea and off west Scotland, there may be a fairly high proportion of birds from Norwegian and Icelandic colonies. However, more data on movements of birds from those colonies would be needed to quantify these proportions accurately.

In the North Sea, gannets in summer show distributions that relate to the locations of breeding colonies (Langston et al. 2013), with birds travelling out from the colony to forage

up to 540 km (and into Norwegian waters) from the colony in the case of the largest colonies such as the Bass Rock (Hamer et al. 2001), predominantly on pelagic fish such as sandeels, herring and mackerel. Foraging ranges from smaller colonies are much shorter. Foraging ranges of gannets breeding in Norwegian colonies are small, which relates at least in part to the small size of those colonies so less competition among foraging adults. Birds equipped with GPS trackers at two Norwegian colonies while breeding fed no more than 22 km from their colony in 2007, no more than 56 km in 2008 and no more than 49 km in 2009 (Pettex et al. 2010) so would not have entered UK waters during their breeding foraging trips. A similar situation probably applies for Faroese and Icelandic breeding gannets. Birds breeding at Irish colonies apparently avoid foraging during the breeding season close to areas used by gannets breeding in UK colonies, so that few gannets in UK waters in summer are likely to be from Irish colonies (Wakefield et al. 2013). However, gannets from the colonies in the Channel Islands apparently forage in UK waters of the western English Channel while breeding (Wakefield et al. 2013).

Many immature gannets, particularly of the 3 and 4 year old cohorts, attend colonies during the summer (mostly from May to August so for a shorter period than breeding adults are present), and those birds tend also to show 'Central Place foraging' with their feeding flights radiating out from the colony, but over larger areas of sea than used by breeding adults (Votier et al. 2011).

7.7 Numbers in UK waters

Tasker et al. (1985) found that about 60% of gannets in the southern North Sea in summer were immatures, whereas in the northern North Sea this was only 20%. In winter, there are few immature gannets in the North Sea (fewer than 7% of all records), and densities of adults are lower than in summer (Tasker et al. 1985). Tasker et al. (1985) reported an average across the North Sea of 0.4 gannets per km^2 in October but only 0.04 per km^2 in December to February. Gannet distribution in the North Sea show a stronger correlation in winter with the distribution of fishing vessels, as they scavenge extensively on trawl fishery discards in winter when pelagic fish are less available (Garthe et al. 1996). Off the west of Britain, gannets were found to be present in relatively much lower numbers in winter than in summer, with gannets in winter mostly associated with fishing vessels (Webb et al. 1990). Surprisingly few occurred within the Irish Sea at any time of year with peak abundance there (in September) still below 0.5 birds per km^2 (Webb et al. 1990). However, large numbers of adults and immatures feed at the shelf-edge in the SW Approaches, in the western English Channel and Celtic Sea in November to February (Stone et al. 1995; White and Reid 1998; Brown and Grice 2005). Forrester et al. (2007) consider that 'a few thousand' may be in Scottish waters during winter, but they define winter as December to February, while also noting that gannets may be back on nest sites from the start of January, whereas lowest numbers at sea in Scottish waters may occur in late November or early December.

7.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the entire species' population, comprising 263,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 390,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 1,160,000 individuals. Birds in UK waters may originate from colonies in UK (255,500 pairs), Ireland (36,000 pairs), Iceland (28,500 pairs), Faroe (2,500 pairs), Norway (4,500 pairs) or Germany (632 pairs) (Mitchell et al. 2004, updated by Wanless et al. 2004, and Dierschke in litt). This gives a biogeographic population with connectivity to UK waters of 327,600 pairs, or 655,000 adults. Associated with this will be about 530,000 immatures, giving a total of around 1,180,000 individuals.



Figure 7.2. Breeding population origins of gannets in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

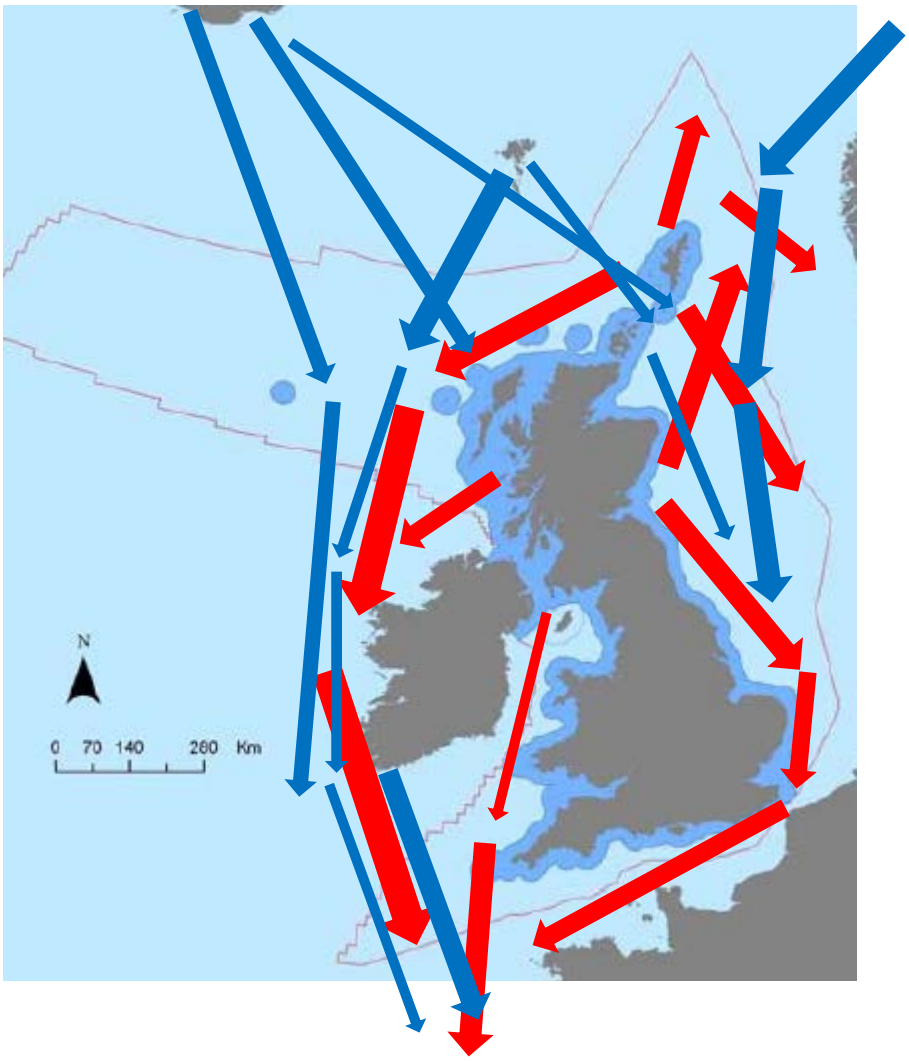


Figure 7.3. Main movements of gannets from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes.

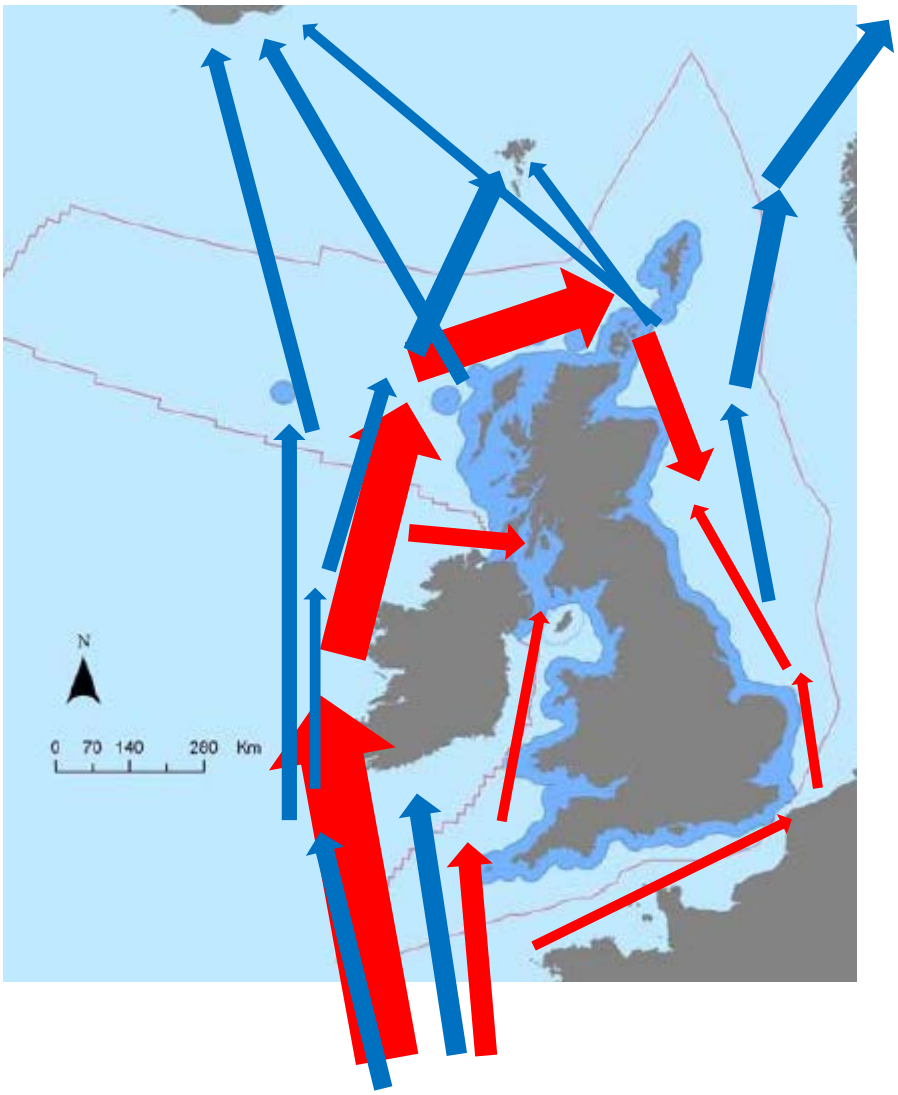


Figure 7.4. Main movements of gannets to UK breeding areas (red arrows) and by overseas populations (blue arrows) through UK waters during 'spring' migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. Note that spring migration routes differ from those in autumn as very few birds migrate through the southern North Sea in spring; most birds returning to colonies in the North Sea do so past the west of Scotland.

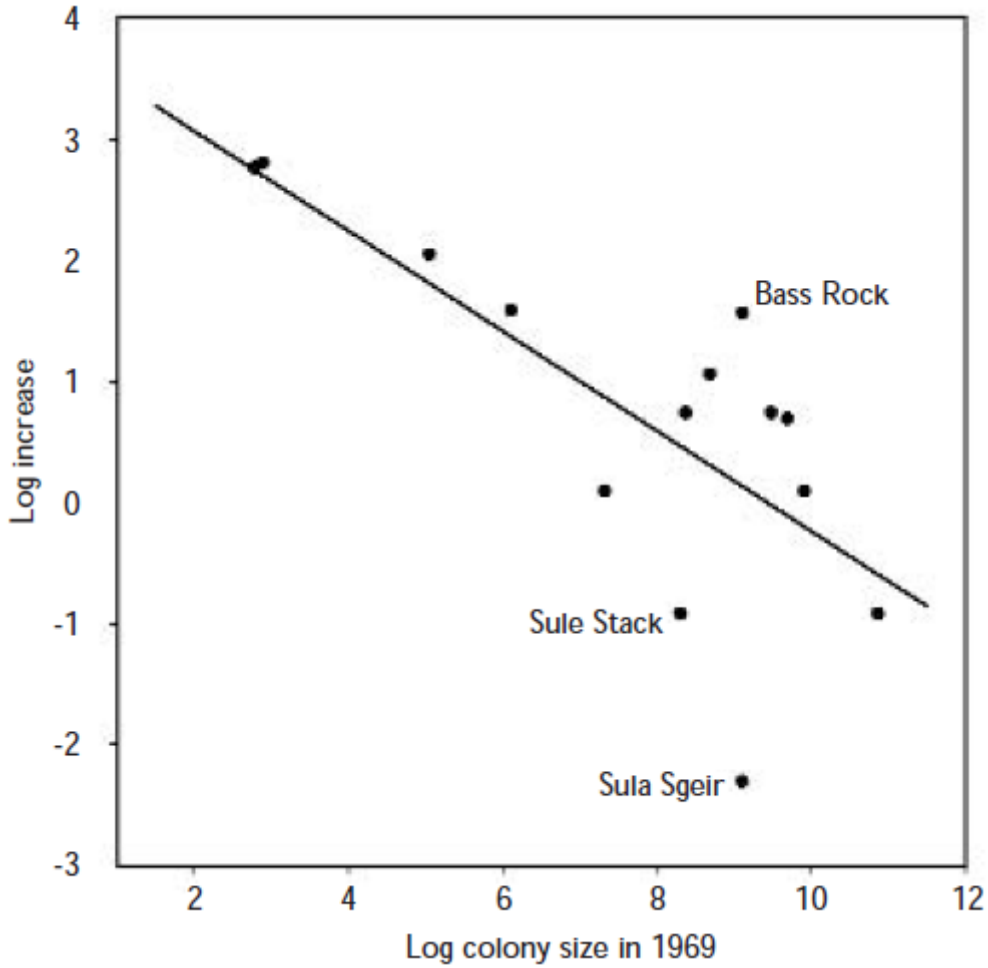


Figure 7.5. Rate of increase in breeding numbers of gannets at each colony in relation to original size of the colony in 1969. Larger colonies grow more slowly. From Wanless et al. (2004). The data are historical but are presented as an example of a pattern that appears to be typical; smaller colonies tend to grow faster than larger colonies, implying density-dependence, probably of recruitment as there is no evidence of reduced productivity in large colonies, and no evidence (though based on very limited data) of differences in adult survival rates between large and small colonies.

7.9 Proportion of UK population from UK breeding SPAs

The 10 SPAs with breeding gannets as a feature designated before 2000 together held 197,127 pairs at designation, estimated to represent ca. 98% of the British breeding population (Stroud et al. 2001). Almost all of these populations have increased in numbers since designation, and smaller colonies have tended to increase more rapidly than the largest colonies (Figure 7.5). Therefore, the proportion of the population in colonies that are not SPAs with gannet as a feature will have increased slightly since designation was completed. Several colonies that are SPAs for seabirds but held too few gannets for that species to qualify as a feature now hold large enough numbers to qualify (Table 7.1). For example, there were 2,787 pairs at Troup, Pennan and Lion's Heads SPA in 2010, 2,760 pairs on the Flannans SPA in 2004, both of which exceed the 1% of UK population threshold numbers based on the current population estimate of 220,000 pairs. However, even with the smaller colonies growing faster than SPA populations, the SPA suite still held 95.9% of the GB population around 2004 (Stroud et al. 2014), and this percentage is likely to remain around 95% in the near future.

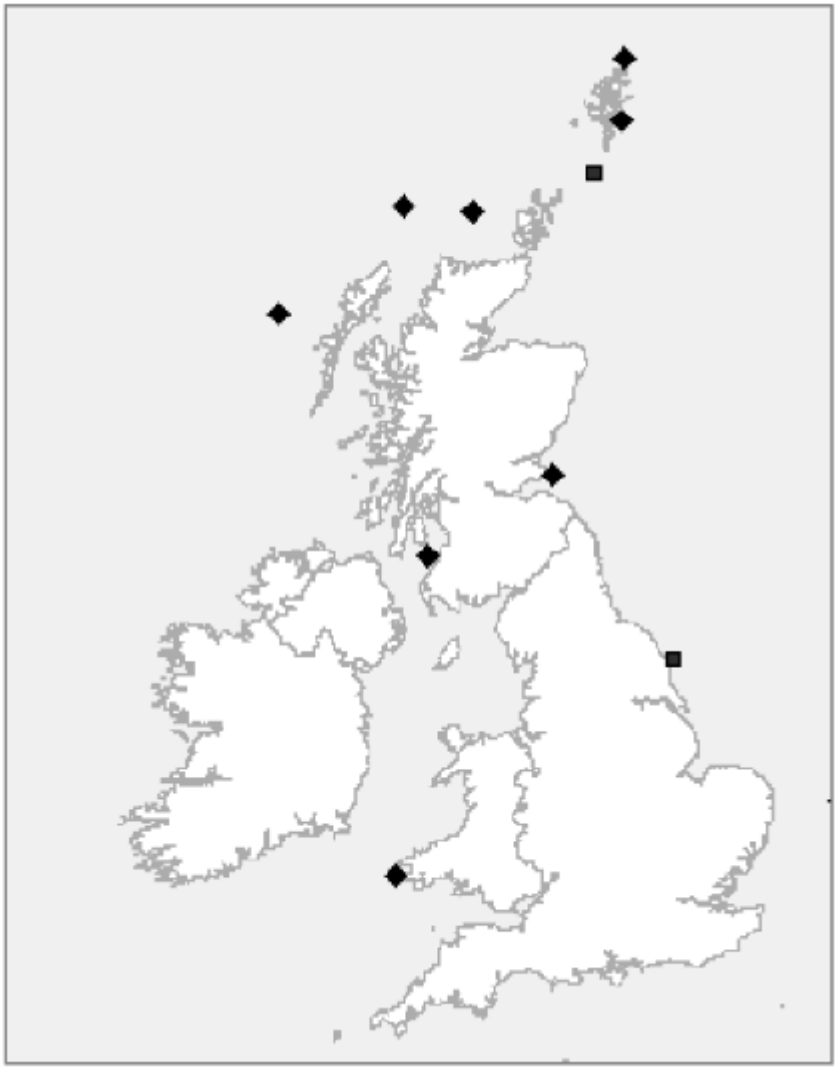


Figure 7.6. The UK SPA suite for gannet. These SPA populations are listed in Table 7.1.

Table 7.1. The UK SPA suite for breeding gannets and data for other major colonies.

SPA	**Location**	**Pairs**	**Year desig-nated**	**Site Condition Monitoring***	**Recent count**	**Year**	**Reference**
UK North Sea & Channel							
Hermaness, Saxavord & Valla	Shetland	12,000 (1994)	1994	Maintained 2008	15,633 24,353	2003 2008	Wanless et al. 2005 Lewis et al. 2012
Noss	Shetland	7,310 (1994)	1996	Maintained 2008	8,652 9,767	2003 2008	Wanless et al. 2005 Lewis et al. 2012
Foula	Shetland	Not yet listed as a qualifying feature			220 280 600 723 919 1,370	1990 1991 1994 2000 2004 2007	SCM database SCM database SCM database SCM database SCM database SCM database

Fair Isle	Shetland	1,166	1994	Maintained 2001	3,968 4,085 3,862 3,924	2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 SCM database SCM database
West Westray	Orkney	Not yet listed as a qualifying feature	1996		167 345 499 583 600 623	2007 2008 2009 2010 2011 2012	SCM database SCM database SCM database SCM database SCM database SCM database
Troup, Pennan and Lion's Heads	NE Scotland	Not yet listed as a qualifying feature	1997		545 1,085 1,228 1,547 1,810 2,787	1995 1998 2001 2004 2007 2010	SCM database SCM database SCM database SCM database SCM database SCM database
Forth Islands	E Scotland	21,600 (1985) Or 34,400 (Stroud et al. 2001)	1990	Maintained 2004	34,397 48,065 55,482	1995 2004 2009	Mitchell et al. 2004 Lewis et al. 2012 Lewis et al. 2012
Flamborough Head & Bempton (to be subsumed into Flamborough and Filey Coast SPA subject to consultation)	E England	2,501 (Stroud et al. 2001)	1993		3,940 3,480 6,487 7,859 11,061	2004 2005 2008 2009 2012	Wanless et al. 2005 SCM database SCM database SCM database SCM database
Flamborough and Filey Coast pSPA	E England	8,469 (2008-2012)	Not yet		As above		As above
UK Western waters							
Sule Skerry and Sule Stack	N Scotland	4,890 (1994)	1994	Maintained 2004	4,675	2004	Wanless et al. 2005
North Rona and Sula Sgeir	N Scotland	10,400 (1994) Or 9,000 (Stroud et al. 2001)	2001	Not reported	10,703 9,225	1999 2004	Lewis et al. 2012 Wanless et al. 2005
St Kilda	Western Isles	50,050 (1985) Or 60,400 (Stroud et al. 2001)	1992	Maintained 2000	60,428 59,622	1995 2004	Mitchell et al. 2004 Wanless et al. 2005
Flannan Isles	Western Isles	Not yet listed as a qualifying feature	1992		414 679 1,438 1,244 2,760	1988 1992 1994 1998 2004	SCM database SCM database SCM database SCM database SCM database

Ailsa Craig	W Scotland	23,000 (1987) or 32,460 (Stroud et al. 2001)	1990	Maintained 2004	32,456 27,130	1995 2004	Mitchell et al. 2004 Wanless et al. 2005
Grassholm	Wales	33,000 (1994)	1986		32,094 39,292	2004 2009	Wanless et al. 2005 SCM database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

7.10 BDMPS

UK gannet numbers are much larger than numbers in Iceland, Norway, Faroe, so that UK birds, almost all of which are from SPA populations, generally predominate throughout UK waters. Gannets migrate southwards after initial autumn dispersal which can be northwards or southwards but birds tend to remain on or at the edge of the continental shelf rather than going into deep oceanic waters. Northern parts of UK waters see a large reduction in gannet numbers from 'autumn' (September-October) into 'winter' (November) and then increasing numbers with return migration in December to March. This could suggest three seasonal divisions: autumn, winter, and spring. However, in southern UK waters there seems to be little evidence of a distinct 'winter' period with low numbers and no migration activity, and numbers recorded monthly at offshore wind farm development sites show little or no winter minumim of numbers, and so it may be more appropriate to define two seasonal periods; 'autumn' (September-November) and 'spring' (December-March). These two migration seasons cannot be aggregated into a single non-breeding period because the migration routes used by gannets are distinctly different in autumn and spring; many birds migrate southwards through UK North Sea waters in autumn, but most migrate northwards in UK western waters in spring, even if returning towards UK North Sea breeding colonies. It makes sense to separate UK North Sea waters from UK western waters as separate BDMPS because the contributions of birds from particular SPA populations differ considerably between these two areas as a result of gannets rarely migrating overland.

The contributions of individual UK SPA populations, UK non-SPA populations, and overseas populations in the four BDMPS (UK North Sea and Channel autumn, UK North Sea and Channel spring, UK western waters autumn, UK western waters spring) are presented in detail in Appendix A Tables 14 to 17.

Based on evidence reviewed in sections 7.5, 7.6 and 7.7, it is estimated that 80% of adults and 80% of immatures from Shetland colonies are in the UK North Sea and Channel autumn BDMPS, as are 100% of adults and 90% of immatures from colonies in eastern Scotland and England, 10% of adults and 20% of immatures from colonies in the northern part of UK western waters (from north Scotland to St Kilda), 0% of adults and 10% of immatures from the southern part of UK western waters (from Ailsa Craig to Wales), 30% of adults and immatures from Iceland, Norway, Faroe, 0% of adults and 10% of immatures from Ireland, and 30% of adults and 40% of immatures from Germany. These sum to a total of 45,174 birds from overseas and 411,125 birds from UK populations, a total of 456,298 overall (Appendix A Table 14).

Based on evidence reviewed in sections 7.5, 7.6 and 7.7, it is estimated that 20% of adults and 10% of immatures from Shetland colonies are in the UK western waters autumn BDMPS, as are 0% of adults and 10% of immatures from colonies in eastern Scotland and England, 90% of adults and 70% of immatures from colonies in the northern part of UK western waters (from north Scotland to St Kilda), 100% of adults and 80% of immatures from

the southern part of UK western waters (from Ailsa Craig to Wales), 20% of adults and 30% of immatures from Iceland, Norway, Faroe, and Ireland, and 0% of adults and 0% of immatures from Germany. These sum to a total of 63,359 birds from overseas and 482,605 birds from UK populations, a total of 545,954 overall (Appendix A Table 15).

Based on evidence reviewed in sections 7.5, 7.6 and 7.7, it is estimated that 70% of adults and 40% of immatures from Shetland colonies are in the UK North Sea and Channel spring BDMPS, as are 70% of adults and 40% of immatures from colonies in eastern Scotland and England, 0% of adults and 0% of immatures from colonies in UK western waters (from north Scotland to Wales), 10% of adults and immatures from Iceland, 20% of adults and immatures from Norway and Faroe, 0% of adults and 10% of immatures from Ireland, and 30% of adults and 30% of immatures from Germany. These sum to a total of 21,903 birds from overseas and 226,482 birds from UK populations, a total of 248,385 overall (Appendix A Table 16). This lower number in the UK North Sea and Channel BDMPS in spring than in autumn reflects the observation that many gannets migrating back towards colonies in the North Sea do so up the west coast of Scotland rather than through the North Sea, so are present in western waters during most of spring migration.

Based on evidence reviewed in sections 7.5, 7.6 and 7.7, it is estimated that 30% of adults and 30% of immatures from North Sea colonies are in the UK western waters spring BDMPS, as are 100% of adults and 80% of immatures from colonies in UK western waters (from north Scotland to Wales), 20% of adults and 20% of immatures from Iceland and Norway, 30% of adults and immatures from Faroe and Ireland, and 0% of adults and 0% of immatures from Germany. These sum to a total of 65,703 birds from overseas and 596,185 birds from UK populations, a total of 661,888 overall (Appendix A Table 17). The higher number in the UK western waters BDMPS in spring than in autumn reflects the observation that many gannets migrating back towards colonies in the North Sea do so up the west coast of Scotland rather than through the North Sea, so are present in western waters during most of spring migration.

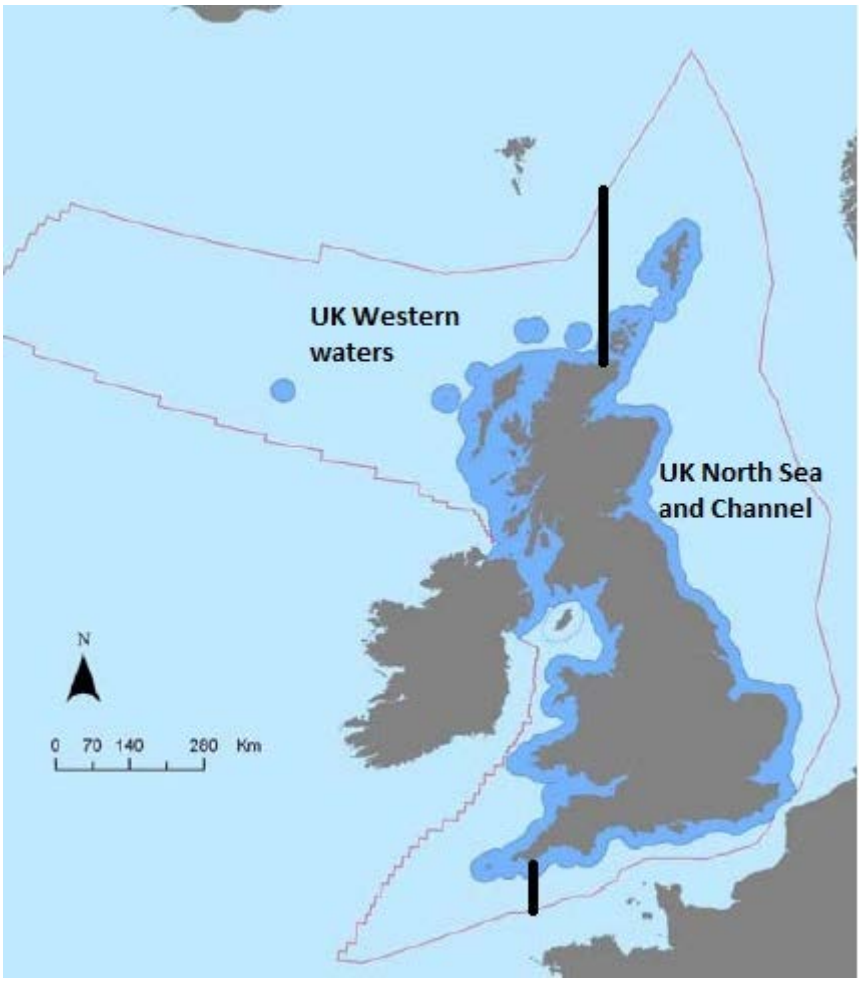


Figure 7.7. Two defined BDMPS spatial areas for gannet; 'UK North Sea and Channel' and 'UK Western waters'.

7.11 Proportions of UK SPA birds in each BDMPS

Since over 95% of UK gannets are in SPA populations, the proportion of UK SPA birds in each BDMPS is virtually the same as the proportion that is from UK colonies. The proportions that are adult SPA birds in each BDMPS total can be computed from data in Appendix A Tables 14-17. For example, in the UK North Sea BDMPS in autumn, there are 208,661 adults from UK breeding gannet SPA populations out of a total of 456,298 birds, giving a proportion of 46% being adults from UK SPA populations.

7.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Since over 95% of UK gannets are in SPA populations, the geographical distribution of UK SPA birds is virtually identical to that of the UK population as a whole. During migrations gannets range widely, and are likely to be thoroughly mixed with birds from other populations across each BDMPS range.

8. GREAT CORMORANT *Phalacrocorax carbo*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (September to March) (adults and immatures)
Overseas	285,000	1,470
UK	39,000	31,653
Total	324,000	33,123

'Non-breeding season' BDMPS (September to March)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
NW North Sea	6,012	98	5,914
SW North Sea & Channel	10,460	1,107	9,353
West of Scotland	7,049	56	6,993
SW England & Wales	9,602	209	9,393

Colour coding is green for numbers from UK colonies in each BDMPS since the locations and sizes of cormorant colonies are well known from survey data and breeding numbers have shown only small changes in total numbers in the UK over recent years, apparently peaking around 2000 and declining slightly since then back to totals similar to those present in the mid-1980s. Colour coding for numbers of cormorants arriving into UK waters from overseas is red in recognition of the fact that the proportions of overseas populations visiting UK are not well known, although numbers of cormorants present in winter have been estimated and indicate that very few overseas birds are present in most of the UK apart from the southern North Sea. Even in the southern North Sea, continental cormorants represent only a small proportion of the total present, considerably outnumbered by UK birds, so that total numbers are mainly determined by the UK numbers, and so are coded green.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 18 to 21.

8.1 Breeding range and taxa

There are six subspecies of great cormorant which is a widely distributed species around the world. However, only two of these subspecies occur in the British Isles; nominate *carbo* breeds in Britain but also the Atlantic coast of Fennoscandia, Iceland and Greenland and breeds mainly at coastal colonies, and *sinensis* breeds mainly at freshwater colonies from northern France to the Baltic Sea and eastwards into China. Although most British and Irish cormorants are of the nominate race *carbo*, some cormorants breeding in Britain and Ireland at freshwater sites are of the continental race *sinensis* (Sellers et al. 1997). There might be potential to identify origins of individual cormorants from biometrics, but this does not seem to have been investigated. Although most cormorants found in UK waters are from the *carbo*

subspecies, substantial numbers of birds of the *sinensis* subspecies visit UK waters on migration and overwinter, these *sinensis* birds being found predominantly in UK southern North Sea waters and being scarce in other parts of the UK marine area.

8.2 Non-breeding component of the population

Great cormorants start to breed when 3 years old (BTO Birdfacts). Adult survival rate is 0.88 (BTO Birdfacts), juvenile survival 0.58 (BTO Birdfacts) and mean productivity is 1.913 chicks per pair (JNCC database, n=62 measurements). To obtain a stable population, survival of immatures was adjusted to 0.5 for juveniles, 0.6 for 1-year olds and 0.7 for 2-year olds. The model population comprised 46% adults, 30% juveniles and 24% older immatures. There are 1.17 immatures per adult.

8.3 Phenology

Breeding colonies are not completely deserted until September (Brown and Grice 2005), but modal departure occurs in late June and July (Pennington et al. 2004; Forrester et al. 2007). However, autumn migration starts in mid-June (Cramp et al. 1977-94), July/August (Wernham et al. 2002; Forrester et al. 2007), or mid-August (Pennington et al. 2004). Peak autumn migration occurs in August-October (Cramp et al. 1977-94), September (Forrester et al. 2007), mid-September (Pennington et al. 2004), September-November (Wernham et al. 2002). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late July and August (Figure 8.1). Autumn migration is completed by early November (Forrester et al. 2007), mid-November (Pennington et al. 2004) or November (Cramp et al. 1977-94; Wernham et al. 2002), excluding a few stragglers still moving in mid-late November.

Spring migration starts in January (Wernham et al. 2002), mid-January (Cramp et al. 1977-94), early March (Pennington et al. 2004) or March (Forrester et al. 2007). Peak spring migration occurs in February-March (Cramp et al. 1977-94; Wernham et al. 2002) or April (Pennington et al. 2004) or April-May (Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late January to late-March (Figure 8.1). Spring migration is completed by early April (Cramp et al. 1977-94), May (Wernham et al. 2002) or late May (Pennington et al. 2004; Forrester et al. 2007).

The first spring records of cormorant in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly from 1 January and the last records were predominantly at 31 December, as cormorants overwinter, while peak autumn migration was reported in August to October in most years, and peak spring migration was reported in March to May in most years. Birds re-occupy breeding sites from February or March, but modal re-occupation occurs in March (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007).

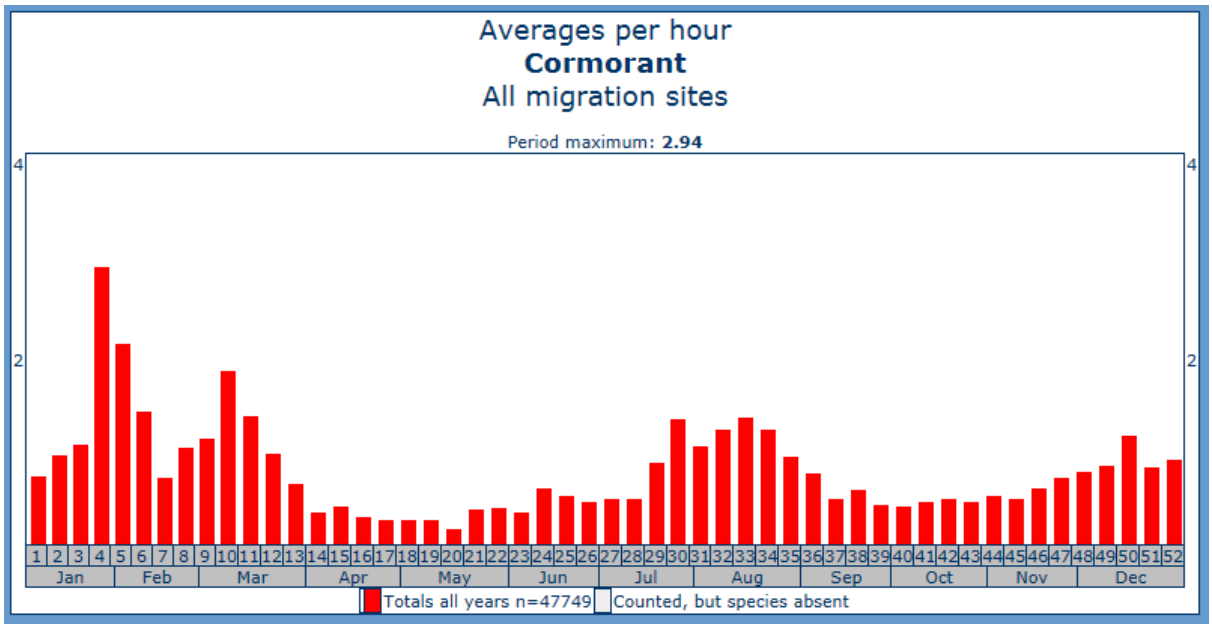


Figure 8.1. Average numbers of great cormorants counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as April-August, non-breeding season September-March.

8.4 Defined seasons:

- UK Breeding season — April-August
- Post-breeding migration in UK waters — August-October (with a few in July and November)
- non-breeding season — September-March (**non-breeding BDMPS**)
- Return migration through UK waters — February-April
- Migration-free breeding season — May-July
- Migration-free winter season — November-January

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for great cormorant:

Non-breeding season BDMPS (September-March).

8.5 Movements of birds from the UK population

Birds from British and Irish coastal colonies mostly overwinter near to their breeding site in coastal habitat (Wernham et al. 2002), but some move onto freshwater habitat in winter (Bearhop et al. 1999). In England, birds show a progressive movement from coastal areas to freshwater sites from September to December (Brown and Grice 2005). The proportion using freshwater habitat in winter has increased (Rehfisch et al. 1999), but is likely to vary according to winter weather, with birds moving back to marine habitats if freshwater sites freeze over. A small proportion of breeders move longer distances south to winter in France or northern Iberia. Long distance movements are more frequent among immatures, especially juveniles (Wernham et al. 2002). Ringing data suggest that birds hardly move south from breeding areas until October, and reach maximum distance south in November, slowly moving northwards from December to May (Wernham et al. 2002). However, it is

evident from observation that birds depart from breeding colonies from July onwards (Wernham et al. 2002) so presumably initial dispersal is of a very limited scale. Almost all adult recoveries are in the breeding area from March onwards, while northward movements of immatures in spring occur later than those of adults (Wernham et al. 2002). There are regional differences around the British Isles in distances and directions moved by cormorants in autumn/winter (Coulson and Brazendale 1968; Wernham et al. 2002). Cormorants from Shetland and Orkney rarely move further south than southern Scotland or northern England. Cormorants from west England may cross the Irish Sea into Ireland, but the predominant direction of movement of those birds is southeastwards into SE England. Cormorants from Wales are the ones most likely to winter inland, predominantly moving to freshwater sites in England. Cormorants from SW England are the ones most likely to winter in Iberia or France. Cormorants tend to be faithful to their particular wintering site; colour ringed birds tend to be observed at their preferred wintering site both within and between winters (Wernham et al. 2002). Most immature cormorants tend to spend the summer close to colonies, although a few may summer in wintering areas.

8.6 Movements of birds from overseas into UK waters

Apart from Irish cormorants moving in small numbers into English freshwater habitat, most foreign-ringed cormorants recovered in the British Isles have been juvenile or immature birds recovered in SE England (where there used to be few breeding colonies). Most recoveries have been in winter, indicating some movement of immatures from continental populations of *sinensis*. These birds, predominantly from the Netherlands, Denmark, or France (Wernham et al. 2002) represent about 2.5% of the British wintering population of cormorants (but 20% of those wintering in freshwater habitat in England), but almost entirely located in SE England. A few of these birds have recruited to breed in SE England. Cormorants ringed as chicks at inland colonies show movement patterns different from UK *carbo* birds and more like those of continental *sinensis* birds (Wernham et al. 2002). Typically, *sinensis* birds migrate further southwards in winter. Many inland reared birds move south into France rather than overwintering in the UK, especially when young. Seabird 2000 reported 8,884 pairs in UK (but numbers have since declined slightly), 4,100 pairs in Ireland, 40,126 pairs in Denmark, 25,150 pairs in Norway, 19,205 pairs in the Netherlands, and 1,500 pairs in France (Mitchell et al. 2004), so populations in Norway, Denmark and the Netherlands are considerably larger than the population in the UK, but only a very small proportion of the birds from those continental populations visit the UK.

8.7 Numbers in UK waters

Highest numbers wintering in marine habitat in English waters occur in coastal areas in NW England (Brown and Grice 2005). However, ESAS data are not informative about cormorant numbers at sea because their distribution tends to be very coastal, in a band that is generally not covered by boat surveys at sea. Forrester et al. (2007) suggested that about 9,000-11,500 cormorants are in Scotland and Scottish waters in winter, and that numbers are not greatly higher during the migration periods. Musgrove et al. (2013) reported that there are 25,000 in Britain in winter, and 41,000 in the UK in winter.

8.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the subspecies *carbo* population, comprising 41,200 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 52,500 pairs. Kober et al. (2010) presented an estimated biogeographic population of 117,900 individuals. In addition, the population of the subspecies *sinensis*, which occurs in small numbers in the UK, is some 300,000 to 330,000 pairs (Brown and Grice 2005). However, numbers of cormorants reaching UK waters from overseas are very small in relation to the large size of the European populations from which they are derived, and represent no more than about 2.5% of the British winter population of cormorants. Almost all of these continental birds occur in SE England, and mostly inland on

freshwater habitat. The biogeographic population with connectivity to UK waters can therefore be defined as the populations of UK (now probably about 8800 pairs so a total of about 39,000 birds including adults and immatures) plus the populations of Denmark, Netherlands, Ireland and France. Those overseas populations sum to 285,000 birds including both adults and immatures. However, it may be more appropriate to consider the total numbers in UK waters in the non-breeding season since very few of those continental birds visit the UK. The total in UK waters in the non-breeding season sums to about 33,500 birds, of which 32,000 originate from UK colonies.



Figure 8.2. Breeding population origins of great cormorants in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

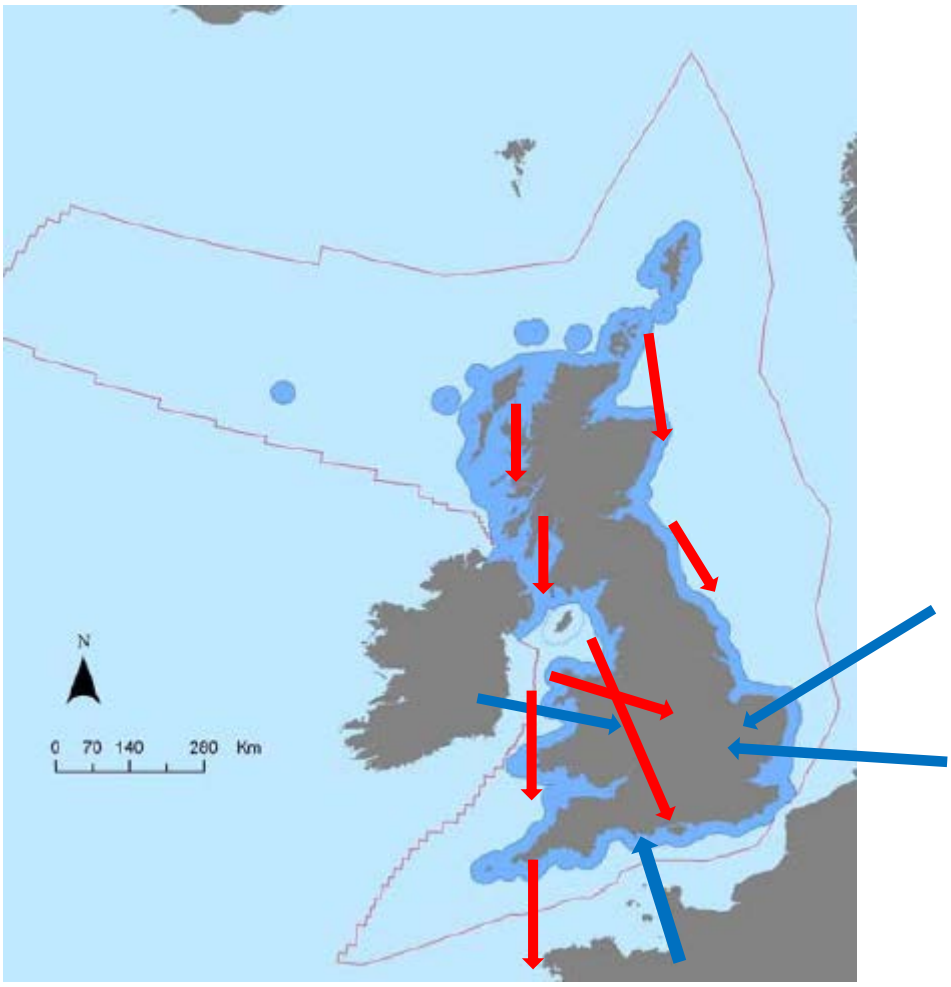


Figure 8.3. Main movements of great cormorants from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

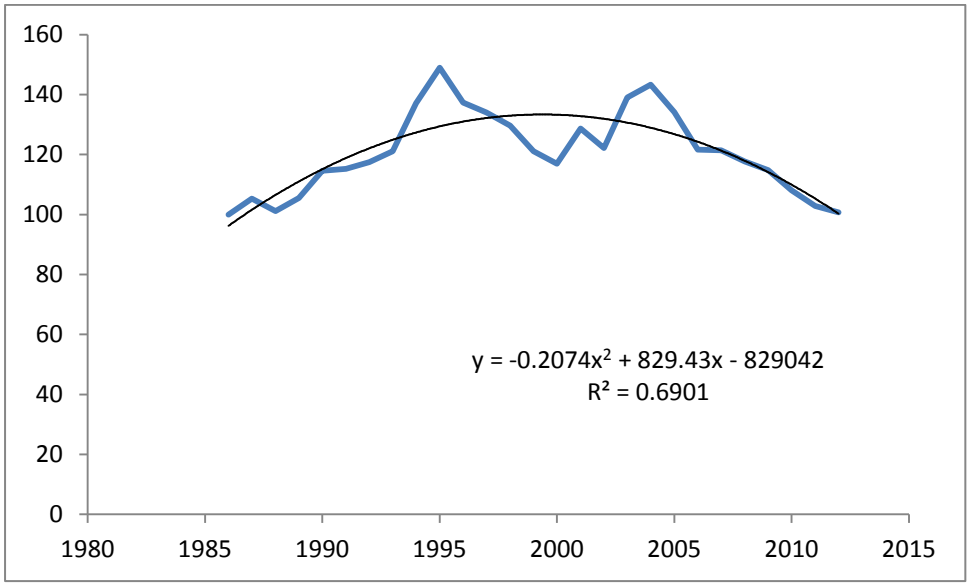


Figure 8.4. Trend in the great cormorant breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

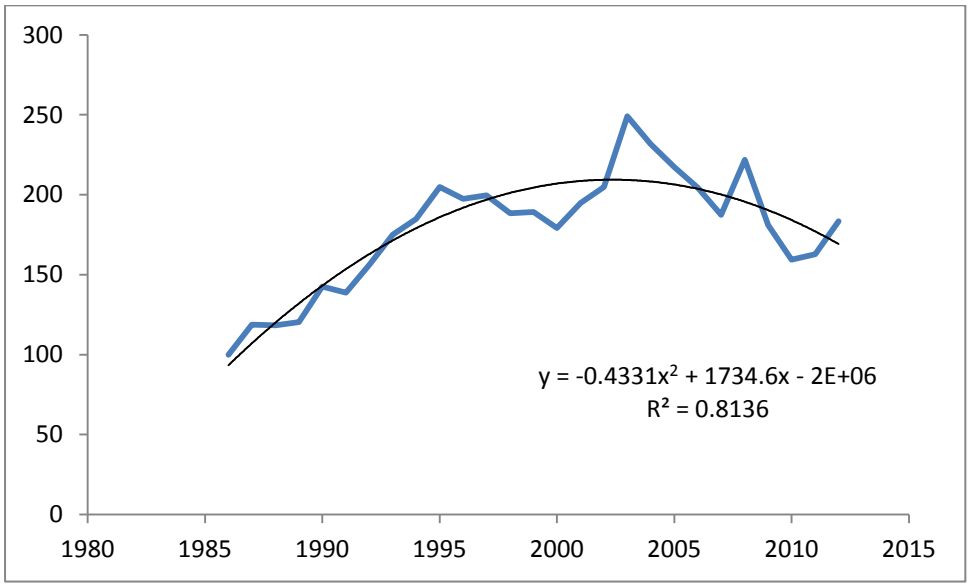


Figure 8.5. Trend in the great cormorant breeding population index in England from 1986-2012. Data from JNCC seabird population monitoring database.

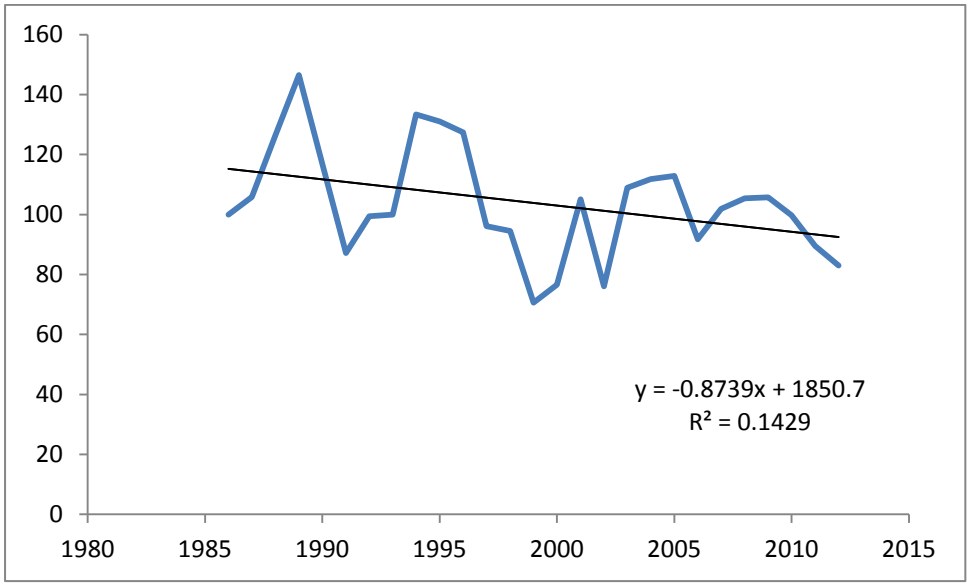


Figure 8.6. Trend in the great cormorant breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

8.9 Proportion of UK population from UK breeding SPAs

The 7 SPAs with breeding great cormorants as a feature together held 2,316 pairs at designation, estimated to represent ca. 30% of the British breeding population (Stroud et al. 2001). Numbers at several of these SPAs have decreased considerably since designation, while the overall population has declined only very slightly relative to numbers around the period of SPA designations. As a consequence the proportion of the GB population breeding within the SPA suite has fallen to an estimated 14.8% in the early 2000s (Stroud et al. 2014). There are also SPAs designated for non-breeding cormorants (Stroud et al. 2001), but those are not relevant in the context of establishing BDMPS.



Figure 8.7. The UK SPA suite for great cormorant. These SPA populations are listed in Table 8.1.

Table 8.1. The UK SPA suite for breeding great cormorants.

SPA	**Location**	**Pairs**	**Year desig-nated**	**Site Condition Monitoring***	**Recent counts**	**Year**	**Reference**
NW North Sea							
Calf of Eday	Orkney	223 (1995)	1998	Maintained 2006	195 204 181	2003 2006 2012	SCM database Lewis et al. 2012 SCM database
East Caithness Cliffs	N Scotland	230 Or 144 (Stroud et al. 2001)	1996	Declined 1999	53 81 67 85 52	2009 2010 2011 2012 2013	SCM database Lewis et al. 2012 SCM database SCM database SCM database
Forth Islands	E Scotland	200 (1985) Or 240 (Stroud et al. 2001)	1990	Declining 2010	102 91 132 57 80	2009 2010 2011 2012 2013	SCM database SCM database Lewis et al. 2012 SCM database SCM database

SW North Sea & Channel							
Farne Islands	NE England	194 (Stroud et al. 2001)	1985		158 145 141 139 121 135 87	2007 2008 2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database SCM database SCM database
Abberton Reservoir	SE England	490 (1993-1997)	1999		370 352 332 322 216	2000 2001 2002 2004 2005	SCM database SCM database SCM database SCM database SCM database
West of Scotland							
Sheep Island	N Ireland	249 (1992-1996)	1992		182 141 100 117 112	2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database
SW England & Wales							
Puffin Island	Wales	556 (1996-2000) Or 776 (Stroud et al. 2001)	2002		383 730 491 606 760 464 484 410 448	2002 2005 2006 2007 2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database SCM database SCM database SCM database SCM database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

8.10 BDMPS

It seems appropriate to define four BDMPS for regions of UK waters (Figure 8.8) based on biological populations present. The 'NW North Sea' region holds about 6,000 cormorants in winter, almost exclusively birds from UK colonies within the NW North Sea region, making the composition of this BDMPS highly distinctive in having predominantly birds from NW North Sea colonies. The 'West of Scotland' region holds about 7,000 cormorants in winter, almost exclusively birds from UK colonies within the West of Scotland region, so again highly distinctive and separate from the other BDMPS populations. The 'SW England and Wales' region holds about 9,600 cormorants in winter, almost exclusively birds from UK colonies, but also including some immature birds from colonies in NW Scotland. The 'SW North Sea and Channel' region holds about 10,500 cormorants in winter, including large numbers of immature birds from colonies in Scotland and small numbers of continental birds. In the SW North Sea and Channel region, many birds move onto freshwater sites during winter, if weather permits. A few thousand birds from the UK population, mostly immatures, winter in France rather than in the UK. Migration of those birds to/from the Continent (mostly northern France) will marginally increase the BDMPS in southern Britain in the migration seasons compared to winter, but this difference is thought to be small enough that the BDMPS can be used for the entire non-breeding period.

Detailed composition of each of these four BDMPS populations is presented in Appendix A Tables 18 to 21.

Based on evidence reviewed in sections 8.5, 8.6 and 8.7, in the UK NW North Sea BDMPS numbers from UK breeding colonies are large enough to provide virtually all of the numbers of cormorants thought to be found in this area in the non-breeding season, so the proportions of overseas populations visiting this area appear to be extremely small. It is estimated that the percentages derived from overseas populations are 0% of birds from Ireland and France, 0.1% of immatures from Denmark, and 0.01% of immatures from The Netherlands, giving an estimate of only 98 birds from overseas populations in this BDMPS (Appendix A Table 18). It is estimated that 100% of adults and immatures from colonies in Orkney and Caithness remain in this BDMPS in the non-breeding season, together with 60% of adults and 50% of immatures from the Forth Islands and 80% of adults and immatures from UK NW North Sea non-SPA colonies, 10% of adults and 20% of immatures from the Farne Islands, 5% of adults and immatures from UK SW North Sea non-SPA colonies, but 0% of birds from Abberton Reservoir. In addition, ringing suggests that small numbers from western waters colonies move into the NW North Sea during the non-breeding season (Wernham et al. 2002) so the proportions are estimated at 0% of adults and 0.1% of immatures from western colonies (Appendix A Table 18). This gives an estimated total of 5,914 birds from UK populations in this BDMPS.

Based on evidence reviewed in sections 8.5, 8.6 and 8.7, in the UK SW North Sea and Channel BDMPS proportions of overseas populations visiting this area appear to be small, but much larger than in the other BDMPS populations. It is estimated that the percentages derived from overseas populations are 0.5% of immatures from Denmark, and 0.1% of adults and 0.1% of immatures from The Netherlands, giving an estimate of 1,107 birds from overseas populations in this BDMPS (Appendix A Table 19). It is estimated that negligible numbers (rounded to 0%) of adults and immatures from colonies in Orkney and Caithness join this BDMPS in the non-breeding season, but that there are 40% of the adults and 50% of immatures from the Forth Islands, and 20% of adults and immatures from UK NW North Sea non-SPA colonies, 90% of adults and 80% of immatures from the Farne Islands, 80% of adults and 70% of immatures from UK SW North Sea non-SPA colonies and from Abberton Reservoir. In addition, ringing suggests that small numbers from western waters colonies move into the SW North Sea during the non-breeding season (Wernham et al. 2002) so the proportions are estimated at 0% of adults and 0.1% of immatures from western colonies (Appendix A Table 19). This gives an estimated total of 9,353 birds from UK populations in this BDMPS.

Based on evidence reviewed in sections 8.5, 8.6 and 8.7, in the UK West of Scotland BDMPS proportions of overseas populations visiting this area appear to be extremely small. It is estimated that the percentages derived from overseas populations are 0.1% of immatures from Ireland and 0.05% of immatures from Denmark, giving an estimate of 56 birds from overseas populations in this BDMPS (Appendix A Table 20). Ring recovery data suggest that 0% of adults and immatures from colonies in the North Sea join this BDMPS in the non-breeding season. Ringing suggests that most birds from colonies in the West of Scotland area remain there during the non-breeding season (Wernham et al. 2002) so the proportions are estimated at 80% of adults and 60% of immatures from Sheep Island, 70% of adults and 50% of immatures from non-SPA colonies (which tend to be further south than Sheep Island so have higher connectivity with the BDMPS to the south of this. Probably a very small proportion of immatures from Welsh colonies may disperse northwards into this BDMPS (Wernham et al. 2002), so this proportion is estimated at 1% (Appendix A Table 20). This gives an estimated total of 6,993 birds from UK populations in this BDMPS.

Based on evidence reviewed in sections 8.5, 8.6 and 8.7, in the UK SW England and Wales BDMPS proportions of overseas populations visiting this area appear to be very small. It is estimated that the percentages derived from overseas populations are 2% of immatures from Ireland, 0.1% of immatures from France, and 0.01% of immatures from Denmark and The Netherlands, giving an estimate of 209 birds from overseas populations in this BDMPS

(Appendix A Table 21). Ring recovery data suggest that 0% of adults and immatures from colonies in the North Sea join this BDMPS in the non-breeding season. Ringing suggests that most birds from colonies in the West of Scotland area remain there during the non-breeding season (Wernham et al. 2002) but that some move south into the UK SW England and Wales BDMPS; the proportions are estimated at 20% of adults and 40% of immatures from Sheep Island, 30% of adults and 50% of immatures from non-SPA colonies (Appendix A Table 21). Although some move south into French waters, many birds from Puffin Island (Wales) and from non-SPA colonies in SW England and Wales remain within this area during the non-breeding season; the proportions are estimated at 60% of adults and 40% of immatures. This gives an estimated total of 9,393 birds from UK populations in this BDMPS.

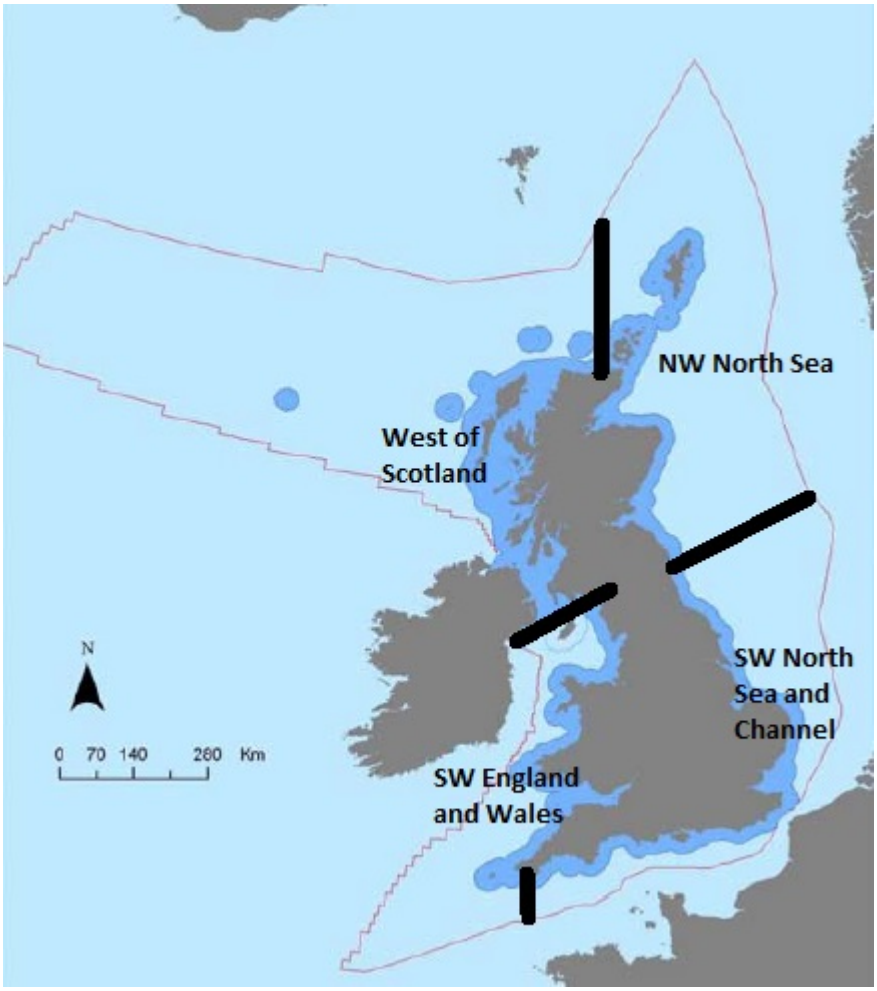


Figure 8.8. Four defined BDMPS spatial areas for great cormorant; NW North Sea, SW North Sea and Channel, West of Scotland, and SW England & Wales.

8.11 Proportions of birds from BDMPS in reference regions

Since almost all cormorants wintering in UK waters are from the UK population and only the SW North Sea and Channel BDMPS receives more than trivial numbers of continental birds, the proportion of birds in each BDMPS that originate from UK SPA breeding populations will be close to the UK average representation of 15%. The NW North Sea region holds the largest number of breeding cormorant SPAs (Table 8.1) but the largest SPA colonies are in the SW England and Wales area and SW North Sea and Channel area. The general population of cormorants breeding in the UK is widely spread across all of these regions, so the proportions of each BDMPS that are birds from UK breeding SPAs will be similar in the four areas. Proportions can be estimated directly from data in Appendix A Tables 18 to 21. For example, for the UK NW North Sea area (Appendix A Table 18), there are estimated to

be 579 adults from UK SPA populations out of a total of 6,012 birds in the non-breeding season BDMPS, giving an estimate of 9.6% of this BDMPS population being adults from UK SPA populations.

8.12 Spatial distribution of UK breeding SPA birds across the BDMPS

The UK breeding cormorant SPA suite is widely distributed across the breeding range of the species in the UK. However, the suite holds only about 15% of the population. Given that many breeding adult cormorants may normally overwinter very close to their breeding site (Wernham et al. 2002 report a median distance between breeding site and wintering site based on ring recovery data of 179 km), it is likely that SPA birds tend to be aggregated in areas close to the seven SPAs, and relatively scarce in areas furthest from the SPAs. However, immature birds are likely to be more widely dispersed than the breeding adults.

9. EUROPEAN SHAG *Phalacrocorax aristotelis*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (September to January) (adults and immatures)
Overseas	9,000	209
UK	97,000	96,078
Total	106,000	96,287

'Non-breeding season' BDMPS (September to January)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
NW North Sea	41,503	0	41,503
SW North Sea & Channel	4,346	0	4,346
West of Scotland	37,363	52	37,311
SW England & Wales	13,075	157	12,918

Colour coding is green for overseas numbers since it is well established from ringing that extremely few shags from overseas populations have ever reached UK waters. Since locations of shag colonies are well known, and shags are known to remain mostly close to their breeding sites throughout the year, colour coding for numbers from UK and total numbers would be green apart from the fact that there is strong evidence for substantial recent declines in numbers at some, but not all, shag colonies. Because some other colonies have not been censused since 1999-2000, there is some uncertainty as to the sizes of those populations (as is evident from Table 9.1 which shows a 90% decline in breeding numbers at Foula SPA in 2000-2013, but much smaller declines at some other sites). This uncertainty seems not enough to code the data red since many of the SPA populations have been counted several times since 2000, and it is likely that declines at non-SPA colonies will be less pronounced since smaller colonies are likely to be less severely affected by density-dependent processes such as competition which is likely to be the cause of declines in numbers.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 22 to 25.

9.1 Breeding range and taxa

The European shag has three subspecies. Nominate *aristotelis* breed from Iceland and northern Scandinavia along the European coast to the Iberian peninsula. *P. a. desmarestii* breeds in the Mediterranean and Black Sea. *P. a. riggenbachi* breeds on the Atlantic coast of

Morocco. Neither of the latter two subspecies has been recorded in UK waters. Biometrics of nominate *aristotelis* do not seem to be useful to identify origins of individual birds.

9.2 Non-breeding component of the population

European shags start to breed when an average of 4 years old (BTO Birdfacts), though Daunt et al. (2003) point out that age of first breeding can vary from 3 to 17 years old in males and 3 to 15 years old in females, while Aebischer et al. (1986) report age of first breeding as 2 for males and 3 for females. Adult survival rate is 0.878 (BTO Birdfacts), juvenile survival 0.38 up to 2 years of age (BTO Birdfacts) and mean productivity is 1.289 chicks per pair (JNCC database, n=237 measurements) (but these can all be greatly affected by weather conditions, especially at exposed colonies on the east coast of Scotland, Frederiksen et al. 2008). To obtain a stable population, survival of immatures was adjusted to 0.5 for juveniles, 0.62 for 1-year olds, 0.72 for 2-year olds, 0.85 for 3-year olds. The model population comprised 43% adults, 28% juveniles and 29% older immatures. There are 1.31 immatures per adult.

9.3 Phenology

Although breeding colonies are not completely deserted until October or November, modal departure occurs in August to October (Pennington et al. 2004; Forrester et al. 2007) and extremely few birds remain at colonies after September. In extreme cases, shags can sometimes still be breeding into October, and the last chicks may not fledge until after October in some years and colonies. However, autumn post-breeding dispersal/migration starts in July (Cramp et al. 1977-94), August (Wernham et al. 2002; Forrester et al. 2007), or mid-August (Pennington et al. 2004). Peak autumn migration occurs in August-October (Cramp et al. 1977-94), September (Pennington et al. 2004), or September-October (Wernham et al. 2002; Forrester et al. 2007). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late August and early September, but autumn passage was not pronounced (Figure 9.1). Autumn migration is completed by late October (Pennington et al. 2004), early November (Forrester et al. 2007) or November (Cramp et al. 1977-94; Wernham et al. 2002).

Spring migration starts in November (Pennington et al. 2004), late November (Forrester et al. 2007), December (Wernham et al. 2002) or mid-January (Cramp et al. 1977-94). Peak spring migration occurs in December (Pennington et al. 2004; Forrester et al. 2007), January (Wernham et al. 2002) or February (Cramp et al. 1977-94). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late November to February, with the most rapid decline in numbers (which may indicate birds returning to breeding areas) in January-March (Figure 9.1). Spring migration is completed by January (Pennington et al. 2004; Forrester et al. 2007), mid-March (Wernham et al. 2002) or mid-April (Cramp et al. 1977-94).

The first spring records of shag in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 1 January and the last records were at 31 December, as large numbers of shags overwinter, while peak autumn migration was reported in August to October in most years, and peak spring migration was not evident in most years. Birds start to re-occupy colonies from the start of January, but modal re-occupation occurs in February (Pennington et al. 2004; Forrester et al. 2007).

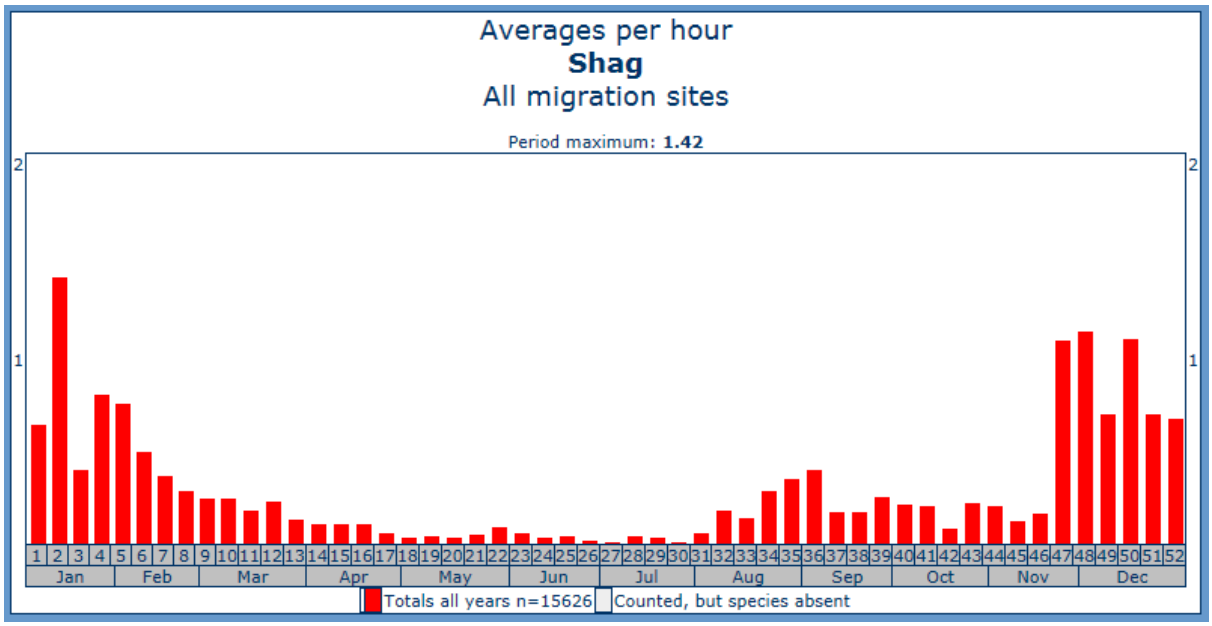


Figure 9.1. Average numbers of shags counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as March-September, non-breeding season October-February. However, from the data reviewed above, a more appropriate definition would be breeding season February-August, non-breeding season September-January.

9.4 Defined seasons:

- UK Breeding season — February-August (sometimes into October)
- Post-breeding migration in UK waters — August-October
- non-breeding season — September-January (**non-breeding BDMPS**)
- Return migration through UK waters — December-February
- Migration-free breeding season — March-July
- Migration-free winter season — November

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for European shag:

Non-breeding season BDMPS (September-January).

9.5 Movements of birds from the UK population

Shags at colonies in the British Isles are considered to disperse and not migrate, and relatively few birds move from their natal colony to breed at another colony (Barlow et al. 2013). However, the extent of dispersal varies between regions (Galbraith et al. 1986), probably to a large extent in response to the ease with which birds can find sheltered areas in the non-breeding season to avoid storms (Harris and Swann in Wernham et al. 2002). Thus birds from colonies in NE England and SE Scotland move the furthest, along a coastline where there is little protection from easterly storms. Indeed, 'wrecks' of shags from colonies in East Britain occur associated with easterly storms (Aebischer 1995), whereas wrecks are very unusual elsewhere in the UK (Frederiksen et al. 2008, Wernham et al. 2002). Birds at colonies in west Britain move very little. Many adults remain within 50 km of

their breeding site throughout the year, even at northernmost colonies (Harris and Swann in Wernham et al. 2002). Immature birds disperse further, on average, than adults (Harris and Swann in Wernham et al. 2002). Very few shags from UK colonies have been recovered outside the UK; a few birds from the northern isles have been recovered in Norway, Denmark and as far as the southern North Sea, and a few from colonies in SW Britain and southern Ireland have been recovered in France (Wernham et al. 2002). Fledglings are fed by parents for some weeks after fledging, and after that period post-fledging dispersal occurs away from colonies. The timing of this dispersal varies greatly as timing of breeding in shags is much earlier in SW Britain than in NE Britain, and the breeding season is very protracted everywhere. So chicks may fledge from April to August.

9.6 Movements of birds from overseas into UK waters

No shags from Norway, Iceland or Faroe have been recovered in the UK (Wernham et al. 2002; Hammer et al. 2013). The only 'foreign-ringed' shags recovered in the British Isles originated from France and the Channel Islands, involving small numbers of birds crossing the English Channel (Wernham et al. 2002). However, some Irish-ringed birds have been recovered in SW England (Brown and Grice 2005) but these are not classified as 'foreign' because Ireland uses the same ringing scheme as the UK. Deployment of geolocators on breeding adult shags at colonies in UK (Isle of May), Iceland (Flatey), and north Norway (Røst and Hornøya) showed that birds from the UK and Icelandic colonies remained close to their colony through the winter. Some birds from Hornøya remained in the Barents Sea near to their colony through winter, but some moved south into the Norwegian Sea (Daunt et al. 2010). However, none of the Norwegian birds moved anywhere near to UK waters. Seabird 2000 reported 26,565 pairs in UK, so even if small numbers of shags from overseas populations occasionally visit UK waters, they are unlikely to represent more than a negligible fraction of the numbers in the UK during migration periods or winter.

9.7 Numbers in UK waters

Shags are not efficiently surveyed by ESAS surveys because they are extremely coastal, and often stand on the shore when not foraging. However, numbers in UK waters will be almost identical to the UK shag population size, since hardly any birds from overseas move into UK waters, and hardly any UK shags move out of UK waters. Numbers of shags in UK colonies have declined considerably since the Seabird 2000 survey, by about 20% from 2000 to 2012 (Figure 9.4), although there are divergent regional patterns with larger decreases in Scotland than in England, and an increase in Wales (Figures 9.5 to 9.7). There are relatively few in Wales though, so the increase there is far smaller than the decrease in Scotland. Overall, the UK breeding population is likely to be about 20,000 to 21,000 pairs now, or up to 42,000 adults. There will be about 55,000 immatures associated with these breeding numbers, so the total population is around 97,000 individuals.

9.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the subspecies *aristotelis* population, comprising 125,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 66,000-73,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 201,800 individuals. Given that movement of birds into and out of UK waters is negligible except with regard to birds from Ireland, an appropriate biogeographic population with connectivity to UK waters would be the UK population of 20,000 to 21,000 pairs, or 97,000 birds including the immatures, plus the population in Ireland of around 2,000 pairs (equivalent to about 9,000 birds including immatures), so a grand total of 106,000 birds. From this population, numbers in the non-breeding season in all UK waters are estimated at 200 birds from overseas, plus 96,000 from UK colonies, giving a grand total in UK waters of 96,200 birds in the non-breeding season.



Figure 9.2. Breeding population origins of shags in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

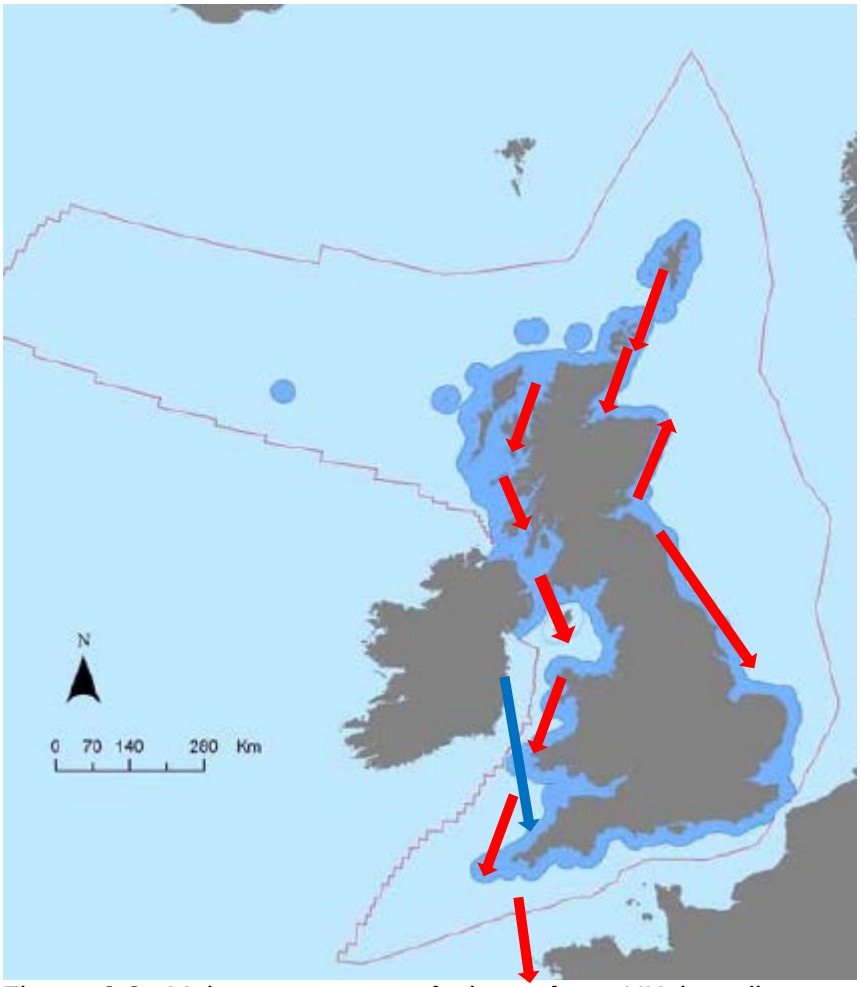


Figure 9.3. Main movements of shags from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration.

Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

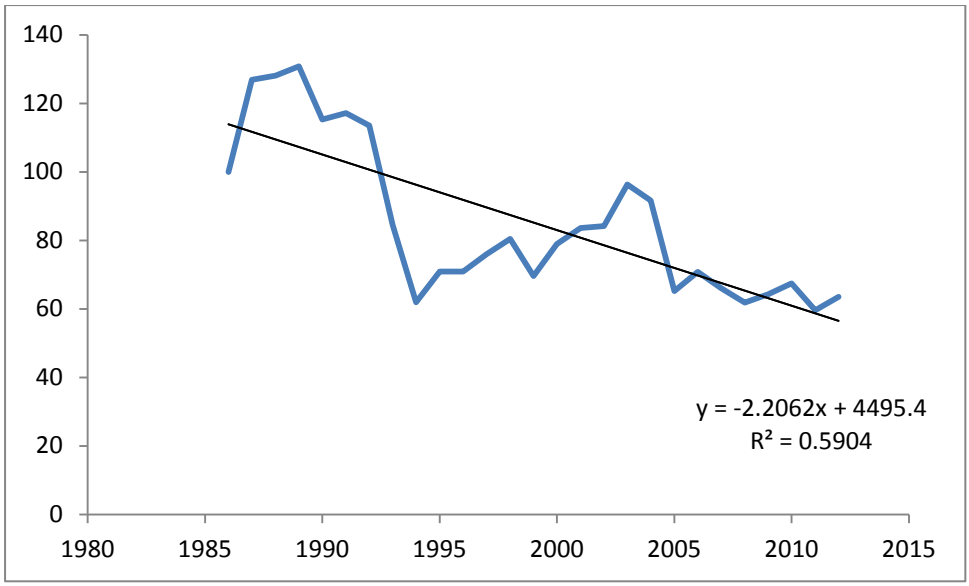


Figure 9.4. Trend in the shag breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

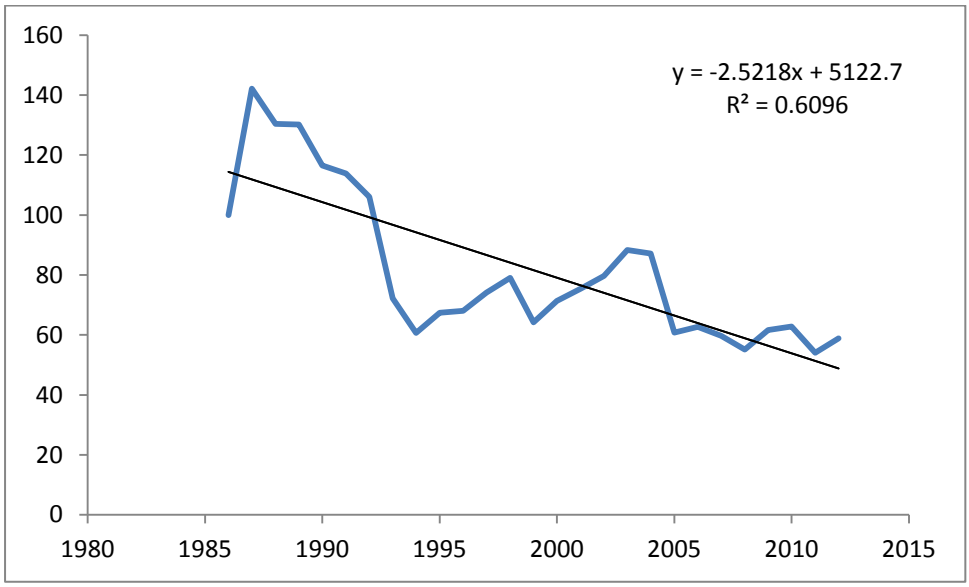


Figure 9.5. Trend in the shag breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

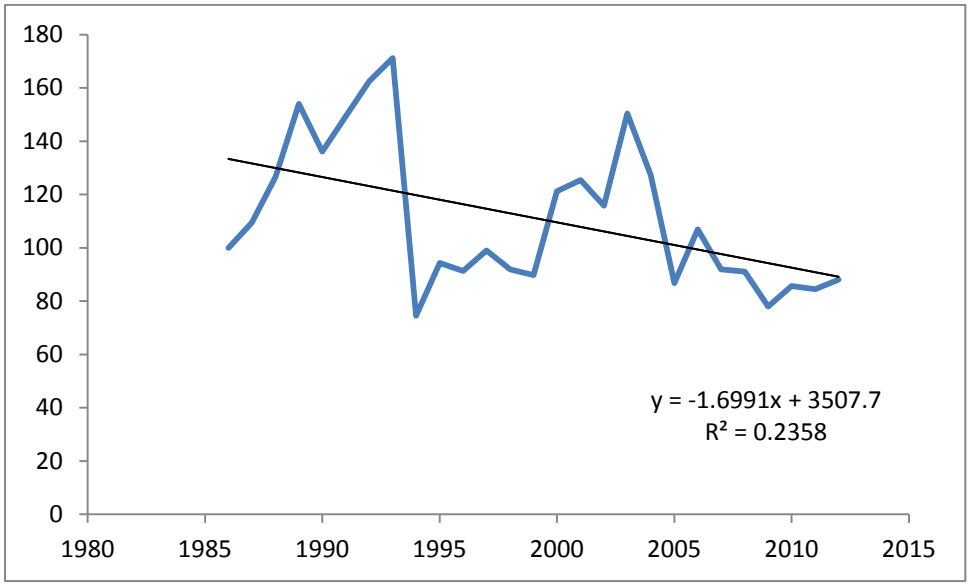


Figure 9.6. Trend in the shag breeding population index in England from 1986-2012. Data from JNCC seabird population monitoring database.

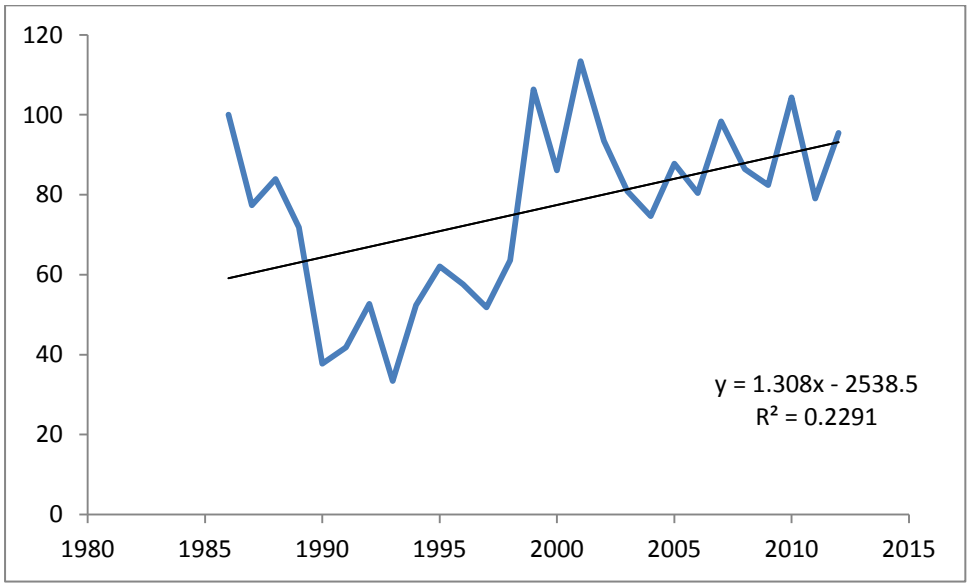


Figure 9.7. Trend in the shag breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

9.9 Proportion of UK population from UK breeding SPAs

The 13 SPAs with breeding shags as a feature together held 17,584 pairs at designation, estimated to represent ca. 47% of the British breeding population (Stroud et al. 2001). Numbers of shags have declined considerably in Scotland, but have declined only slightly in England and have increased slightly in Wales (but because most shags in the UK breed in Scotland, the better performance further south does not compensate for declines in Scottish colonies). Some colonies have declined very dramatically (for example the largest colony in Europe was at Foula, Shetland, and that fell from around 3,000 pairs in the 1970s to 2,277 pairs in 2000, and fewer than 200 pairs in 2013. Many of the largest declines appear to have occurred at the largest colonies, consistent with a density-dependent impact of reduced food supply. As a consequence, the proportion of the population within the SPA suite for shags fell to about 34% of the GB population in the 2000s (Stroud et al. 2014). The proportion within the SPA suite has almost certainly fallen further still since then (for example Stroud et

al. 2014 used the 2000 estimate of 2,300 pairs for Foula whereas now that number is down to <200). The suite probably now holds around 25-30% of the UK shag population.

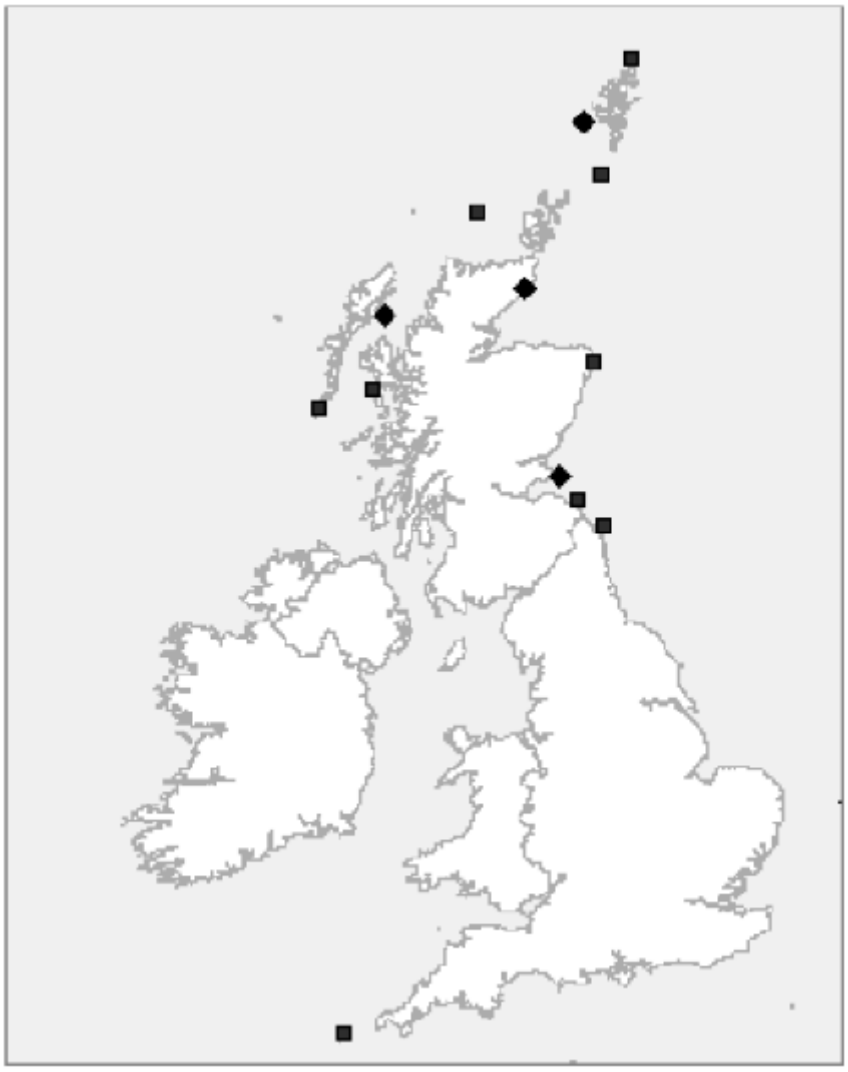


Figure 9.8. The SPA suite for shag. These SPA populations are listed in Table 9.1.

Table 9.1. The UK SPA suite for breeding shags.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent counts	Year	Reference
NW North Sea							
Hermaness, Saxavord & Valla Field	Shetland	540	1994	Declined 2002	82 H'ness only: 94 33 41	1999 1994 1999 2002	Stroud et al. 2014 SMP database SMP database SMP database
Foula	Shetland	2,400 (1997)	1995	Declined 2007	2,300 258 <200	2000 2007 2013	Seabird2000 SMP database Gear 2013
Fair Isle	Shetland	1,099	1994	Declined 2008	567 663 732 235 204	1998 2001 2003 2008 2013	SMP database SMP database SMP database Lewis et al. 2012 SMP database

East Caithness Cliffs	N Scotland	2,345 (1986)	1996	Declined 1999	1,056	1999	Seabird2000
Buchan Ness to Collieston Coast	NE Scotland	1,045	1998	No change 2007	344 331	2007 2007	Lewis et al. 2012 Stroud et al. 2014
Forth Islands	E Scotland	2,400 (1985) Or 2,887 (Stroud et al. 2001)	1990	Recovering 2001	1,088 1,050 1,060 850	2010 2011 2012 2013	Lewis et al. 2012 SMP database SMP database SMP database
SW North Sea & Channel							
St Abb's Head to Fast Castle	E Scotland	651	1997	Declined 2008	329 269 160	2000 2000 2011	Stroud et al. 2014 Seabird2000 Lewis et al. 2012
Farne Islands	NE England	994 (Stroud et al. 2001)	1985		1,059 1,015 838 925 926 965 582	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
West of Scotland							
Sule Skerry and Sule Stack	N Scotland	874 (1986)	1994	Maintained 1998	701 724 15 200	1993 1998 2007 2011	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012
Shiant Isles	Western Isles	1,780 (1986)	1992	Maintained 1999	506	1999	Seabird2000
Canna and Sanday	Inner Hebrides	1,140	1998	No change 2006	305 226 270 255	2010 2011 2012 2013	SMP database SMP database SMP database SMP database
Mingulay and Berneray	Western Isles	721 (1985)	1994	Declined 2009	281 330 115	1998 2003 2009	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012
SW England & Wales							
Isles of Scilly	SW England	1,108	2001		1,296	2006	SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

9.10 BDMPS

Since adult shags show only very limited migration (most adults recovered in the non-breeding season being within 50 km of their breeding site; Wernham et al. 2002), UK waters can be split into several distinct non-breeding season BDMPS for shags. Birds from North Sea colonies tend to be more mobile than birds from western waters colonies, probably due to the greater exposure of east coast waters compared to relatively sheltered conditions in much of the west coast coastline. Based on evidence reviewed in sections 9.5, 9.6 and 9.7, the UK NW North Sea region holds about 41,500 birds in winter, with some birds, especially immatures, moving up or down much of the coastline. The West of Scotland region holds about 37,000 birds in winter, almost all derived from local colonies in that area. The SW England and Wales region holds about 13,000 birds in winter, many of which are immature birds from breeding sites further north, as breeding numbers in that region are relatively

small but immatures from colonies further north move southwards into the area with relatively few locally breeding birds. The SW North Sea and Channel holds about 4,000 birds in winter, most of which are immature birds from breeding sites further north. Numbers during migration periods are essentially the same as these wintering numbers, so the BDMPS are appropriate for migration periods as well as wintering period.

The UK NW North Sea BDMPS has no birds from overseas populations. All adults from colonies in Shetland to Berwickshire are likely to remain within this BDMPS in the non-breeding season. All immatures from Shetland to Aberdeenshire are also likely to remain in the area, while it is estimated that 90% from Forth Islands and 80% of immatures from St Abbs Head area do so. It is estimated that 30% of adults and 40% of immatures from the Farne Islands spend the non-breeding period in the UK NW North Sea BDMPS. No birds from western colonies are thought to move into the area during the non-breeding season so that connectivity with populations to the west of the UK is negligible or zero. These figures result in an estimated BDMPS population of 41,503 birds in the UK NW North Sea BDMPS (Appendix A Table 22).

There have been a few recoveries of ringed shags from NW France in SE England (Wernham et al. 2002), but these appear to be negligible numbers from a small population in which many birds have been ringed, so connectivity between the French breeding population and UK waters is considered to be negligible. On this basis, the UK SW North Sea and Channel BDMPS has no significant numbers of birds from overseas populations. Although no birds from colonies in Shetland to Aberdeenshire are likely to move into the UK SW North Sea and Channel BDMPS, it is estimated that 10% of immatures from Forth Islands and 20% of immatures from St Abbs Head area do so. It is estimated that 70% of adults and 60% of immatures from the Farne Islands, and all birds from the non-SPA colonies in UK SW North Sea and Channel spend the non-breeding period in the UK SW North Sea and Channel BDMPS. No birds from western colonies are thought to move into the area during the non-breeding season so that connectivity with populations to the west of the UK is negligible or zero. These figures result in an estimated BDMPS population of 4,346 birds in the UK SW North Sea and Channel BDMPS (Appendix A Table 23).

Based on evidence reviewed in sections 9.5, 9.6 and 9.7, the UK West of Scotland waters BDMPS has small numbers of birds from Irish populations; it is estimated that perhaps 1% of immatures from Ireland spend the non-breeding season in this BDMPS (an estimated 52 birds). No birds from North Sea colonies are likely to be in this BDMPS in the non-breeding season. All birds from colonies in west Scotland are thought to remain within the area during the non-breeding season, but no birds from Wales and SW England are thought to move into the area. These figures result in an estimated BDMPS population of 37,311 birds in West of Scotland waters BDMPS (Appendix A Table 24).

Based on evidence reviewed in sections 9.5, 9.6 and 9.7, the UK Wales and SW England waters BDMPS has small numbers of birds from Irish populations; it is estimated that 3% of immatures from Ireland (157 birds) are in the BDMPS in the non-breeding season. No birds from North Sea or West of Scotland colonies are thought to move into the area during the non-breeding season. All birds from the Isles of Scilly and from non-SPA colonies in SW England and Wales are thought to remain within this BDMPS in the non-breeding season. These figures result in an estimated BDMPS population of 13,075 birds in the UK Wales and SW England waters BDMPS, with 12,918 coming from UK colonies (Appendix A Table 25).

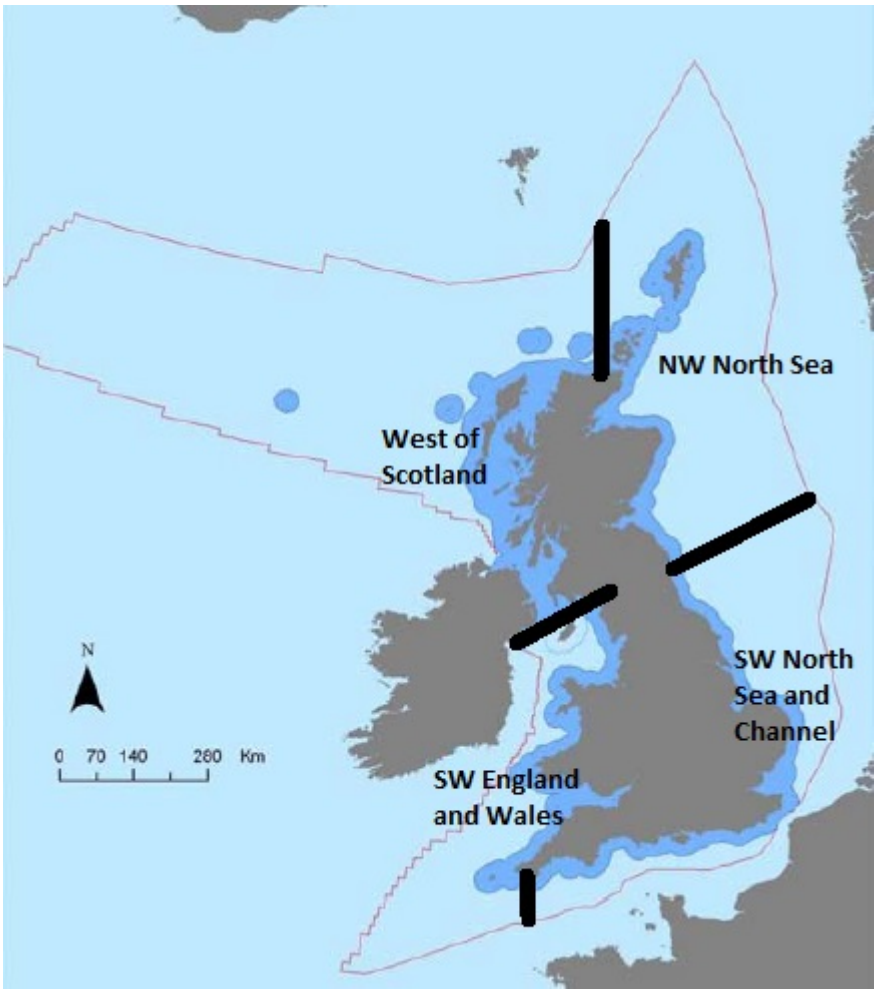


Figure 9.9. Four defined BDMPS spatial areas for shag; NW North Sea, SW North Sea and Channel, West of Scotland, and SW England & Wales.

9.11 Proportions of UK breeding SPA birds in each BDMPS

The distribution of breeding shag SPA populations is closely similar to the overall distribution of breeding shags in the UK. While almost all of the SPA sites are in the northern BDMPS (with the sole exception of the Isles of Scilly), most shags occurring in winter in the SW North Sea and Channel are immature birds dispersed from sites in the NW North Sea, so include immatures from SPAs. However, there are no breeding shag SPA populations in Wales or SW Scotland where there are breeding colonies, so the proportion of SPA birds in the SW England and Wales BDMPS will be lower than in the others. Proportions of adults from SPA colonies in each BDMPS can be computed from data in Appendix A Tables 22 to 25. For example, in the UK NW North Sea non-breeding season BDMPS, there are estimated to be 41,503 birds, of which 6,033 are adults from SPA populations, so those birds represent 14.5% of the total present in that BDMPS.

9.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Although only about 25-30% of shags in UK waters are from SPA populations, the 13 SPAs with breeding shags as a feature are well distributed across the breeding range of this species in the UK. Because adult shags may remain at colony sites through the winter, there is likely to be a tendency for SPA birds to be aggregated close to SPAs at all times of year (ring recoveries suggest that most adults remain within 50 km of their breeding area during the non-breeding season; Harris and Swann in Wernham et al. 2002). This aggregation may be most evident in the West of Scotland and SW England and Wales regions, where shags are most sedentary (Harris and Swann in Wernham et al. 2002). Birds from SPAs in the NW

North Sea region tend to disperse further. However, to counteract that effect, there are relatively more small non-SPA colonies of shags in the West of Scotland region between the SPA sites. Clearly if most adults move only a few tens of kilometres between breeding sites and wintering sites, the shags from colonies in on part of a BDMPS will not mix extensively with shags from areas on the other end of the BDMPS area. It might therefore be appropriate in assessments of impacts to define a reference area smaller than an entire BDMPS centered around a development site, and focus on the populations within that defined reference area. An appropriate reference area might be smaller in UK western waters than in the North Sea since shags are less mobine in western waters than in North Sea waters. Which populations should be included can be assessed from data presented in Tables 22 to 25. It would probably be appropriate to consider birds from all colonies within a radius of 300 km from a development site, but exclude consideration of birds from colonies at greater distances (since ring recoveries even of immature birds are predominantly from within 100 km of the location where the bird was originally ringed).

10. ARCTIC SKUA *Stercorarius parasiticus*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in autumn (August to October) (adults and immatures)	Numbers in UK waters in spring (April-May) (adults and immatures)
Overseas	226,000	9,064	3,786
UK	3,000	2,650	2,552
Total	229,000	11,714	6,338

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Autumn migration BDMPS (August to October)			
UK North Sea and Channel	6,427	5,216	1,211
UK Western waters	5,287	3,848	1,439
Spring migration BDMPS (April-May)			
UK North Sea and Channel	1,227	582	645
UK Western waters	5,111	3,204	1,907

Although there are relatively few colonies of Arctic skuas in the UK, and the species is relatively easy to census, the numbers breeding in UK colonies have declined dramatically in recent years, with this species moving directly from being Green-listed to Red-listed as a consequence of the large decrease in breeding numbers. In addition, several colonies have not been censused since Seabird2000, so that current numbers are uncertain, especially in areas where the species is widely scattered at low density – areas where population trends may differ from those at large colonies with high nesting density. However, most SPA populations have been counted several times since 2000, and a complete survey was carried out in Orkney in 2010. So estimated numbers of UK birds migrating through UK waters are coded amber. Numbers of Arctic skuas that pass through UK waters have been estimated from sources such as seawatching data and ESAS data and reported in several publications (e.g. Forrester et al. 2007), but these numbers are relatively uncertain, and seem to vary from year to year, especially during spring migration when passage is predominantly west of the UK and may be more evident in years when weather conditions

bring birds closer to land. Therefore, total numbers in BDMPS are coded red. Many of the birds passing through UK waters are from overseas populations rather than UK populations and although colour phase data can provide some indication of the origins of Arctic skuas, numbers that originate from overseas populations are rather uncertain, so are also coded red.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 26 to 29.

10.1 Breeding range and taxa

The monotypic Arctic skua is a trans-equatorial migrant and the UK is at the extreme southern limit of its breeding range which is circumpolar and largely Arctic (Furness 2010). Although there is no evidence that biometrics can be used to identify origins of individuals, Arctic skuas have two colour phases, with clinal variation in the proportions. Dark birds predominate at colonies at the southern edge of the range whereas all birds at high Arctic breeding sites are pale phase birds.

10.2 Non-breeding component of the population

Arctic skuas start to breed when 4 years old (BTO Birdfacts). Adult survival rate is 0.886 (BTO Birdfacts), juvenile survival 0.68 (BTO Birdfacts) and mean productivity is 0.522 chicks per pair (JNCC database, n=82 measurements). This estimate of productivity is low, but is certainly representative of breeding performance in the UK in recent decades. Productivity may be higher than this in regions where populations are performing better. However, for the population model, using a low value of productivity tends to be compensated for by increased estimates of juvenile and immature survival in order to achieve a stable population, so the exact value used in the model does not greatly alter the estimated proportion of immatures per adult. To obtain a stable population, survival of immatures was adjusted to 0.69 for juveniles, 0.8 for 1-year olds, and 0.886 for older age classes. The model population comprised 58% adults, 15% juveniles and 27% older immatures. There are 0.71 immatures per adult.

10.3 Phenology

Breeding colonies in the UK are deserted in August, with modal departure in early August (Pennington et al. 2004; Forrester et al. 2007). Autumn migration starts in early August (Wernham et al. 2002; Pennington et al. 2004) or August (Cramp et al. 1977-94; Forrester et al. 2007). Peak autumn migration occurs in August-September (Wernham et al. 2002; Pennington et al. 2004), early September (Forrester et al. 2007), September in English waters (Brown and Grice 2005), or September-October (Cramp et al. 1977-94) (but this last includes migration through southern hemisphere waters). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred very distinctly in late-August and early-September (Figure 10.1). Autumn migration is completed by late October (Pennington et al. 2004; Forrester et al. 2007) or late-November when also considering continued migration through southern hemisphere waters (Cramp et al. 1977-94).

Spring migration starts in late-March from southern hemisphere wintering areas (Cramp et al. 1977-94) but birds start to reach UK waters in early April (Wernham et al. 2002) or April (Pennington et al. 2004; Forrester et al. 2007). Peak spring migration occurs in April-May (Cramp et al. 1977-94; Wernham et al. 2002), early May (Pennington et al. 2004) or May (Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late April and early May (Figure 10.1). Spring migration is completed by late May (Cramp et al. 1977-94; Pennington et al. 2004), early June (Wernham et al. 2002) or June (Forrester et al. 2007).

The first spring records of Arctic skua in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were between 10 April and 7 May, but mostly in mid-April. The last records in autumn fell between 3 September and 8 November but mostly in October. Peak autumn migration was reported in July-September in most years, and peak spring migration was reported in May in most years. Birds re-occupy colonies from early April, with modal return in late April (Pennington et al. 2004; Forrester et al. 2007).

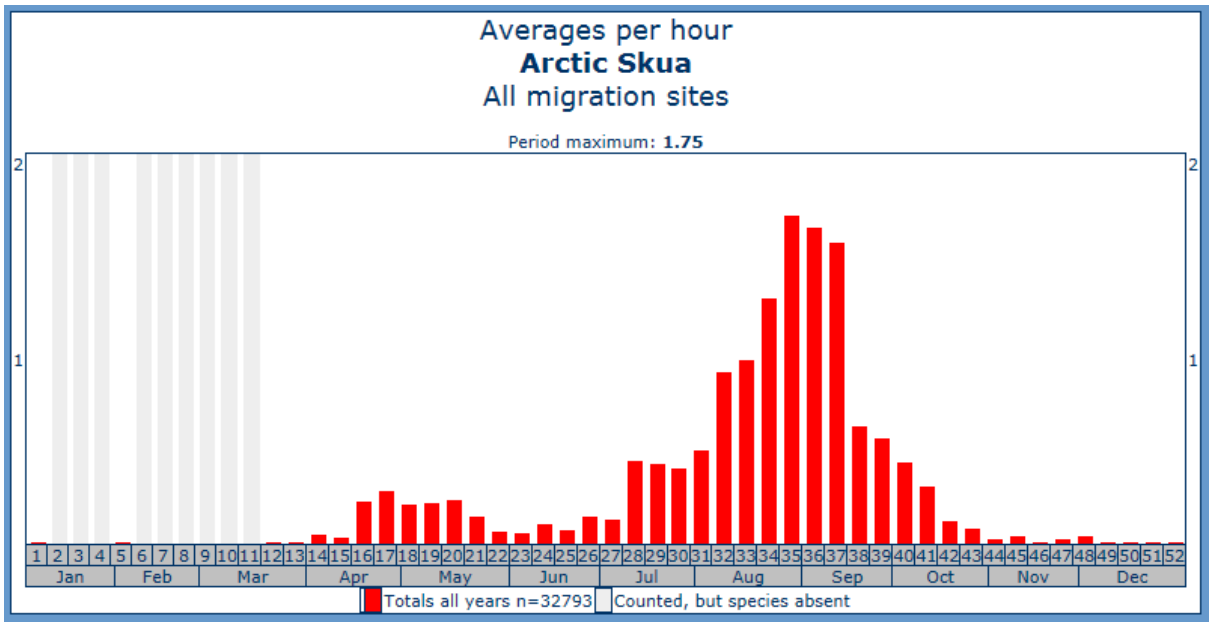


Figure 10.1. Average numbers of Arctic skuas counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season September-April. However, from the data reviewed above, a more appropriate definition would be breeding season May-July, non-breeding season August-April.

10.4 Defined seasons:

- UK Breeding season — May-July
- Post-breeding migration in UK waters — August-October (**autumn BDMPS**)
- non-breeding season — August-April
- Return migration through UK waters — April-May (**spring BDMPS**)
- Migration-free breeding season — June-July
- Migration-free winter season — November-March

Apart from the breeding season, two seasonal BDMPS periods are considered to be appropriate for Arctic skua:

'Autumn' (post-breeding) migration BDMPS (August-October); and

'Spring' (pre-breeding) migration BDMPS (April-May).

10.5 Movements of birds from the UK population

Some failed breeders and some immatures attending UK colonies as pre-breeders may set off on autumn migration as early as July, but most fledglings and adults at UK colonies

depart in early August (Furness 2010; Wernham et al. 2002). Birds from North Sea colonies (Orkney and Shetland) disperse in autumn either through the North Sea or through western waters. Birds from colonies in western waters probably disperse through western waters mainly southwards or southwestwards rather than moving into the North Sea. However, spring migration seems to be more often through western waters, even for adults returning to colonies within the North Sea (Orkney, Shetland and Caithness).

10.6 Movements of birds from overseas into UK waters

Migrants from populations further north pass through British waters mainly in August-September. Autumn migration tends to occur close to the coast. At this time, individuals may hang around areas where there are flocks of terns. A few stragglers may still be present in October, but records from November are extremely scarce (and may involve identification errors as pomarine skuas may occasionally still be seen in November). No Arctic skuas overwinter in British waters. Return migration in spring tends to be more rapid, and with a high proportion of birds passing up the west side of Scotland rather than through the North Sea (Forrester et al. 2007). The proportion of light phase birds tends to increase through spring, as birds that breed at more southerly colonies (where dark phase birds predominate) tend to arrive first, with birds travelling on to the Arctic (where virtually all birds are pale phase) migrating later (Newnham 1984). Scottish adult Arctic skuas return to colonies in late April and May, but Arctic-breeding individuals may not occupy breeding grounds until June (Wernham et al. 2002). It is during May that the proportion of dark phase Arctic skuas is lowest in UK waters, consistent with these birds being predominantly from northern populations (Tasker et al. 1987). There are around 8,000 pairs in Fennoscandia, 7,500 pairs in Iceland, 750 pairs in the Faroes, and tens to hundreds of thousands of pairs on the Arctic tundra bordering the North Atlantic (Mitchell et al. 2004); figure of 50,000 pairs has been used in this report but that estimate is fairly uncertain. Small proportions of each of those populations are thought to migrate through UK waters, but there is very little evidence to indicate which of those populations predominate in the migration season.

10.7 Numbers in UK waters

Autumn migration of Arctic skuas in English waters is seen especially off the coast of E England, whereas spring migration is mainly seen off the S coast and rather few pass along the coast of E England (Brown and Grice 2005). In spring, numbers moving north along the east coast of Scotland tend to be small, but there can be large numbers off the west of Scotland, although these may often pass too far from the coast to be seen from land. As a result, numbers migrating through UK waters are not well defined, but Forrester et al. (2007) suggest that spring migration involves around 1,000 to 5,000 birds in Scottish waters, predominantly to the west of Scotland, while autumn migration involves 1,000 to 10,000 birds, with possibly slightly more than half of these off the west coast, but much better data on numbers available from observations at the east coast. These numbers are likely to be underestimates of the strength of migration of this species, particularly because the species is easily overlooked during boat-based surveys, and because migration can occur in pulses of birds passing beyond sight from shore-based observation points unless driven inshore by weather.

10.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the NE Atlantic population, comprising 30,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 15,000-35,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 75,000 individuals. It is likely that most of this widely distributed biogeographic population has connectivity with UK waters, but that the proportion of the population passing through UK waters is rather small. The UK population of Arctic skuas is small. Seabird 2000 recorded 2,136 AOTs (approximately equivalent to pairs) and numbers have declined considerably since 2000; data presented by Foster and Marrs (2012)

suggest a 57% decline in numbers of AOTs at monitored colonies between 2000 and 2011, so the current UK population may be around 1,000 AOTs. However, the decline in numbers of AOTs does not necessarily mean a proportionate decline in population size, since adults from many of the abandoned AOTs may simply be non-breeding during times of low food supply, and might reoccupy AOTs if conditions were to improve. The biogeographic population with connectivity to UK waters is therefore estimated at 3,000 birds from the UK population and 226,000 birds from overseas populations, giving a total of 229,000 but with a very high uncertainty associated with this estimate. Total numbers in UK waters during autumn migration are estimated at 9,000 birds from overseas and 2,600 from UK populations, so about 12,000 birds overall. Total numbers in UK waters during spring migration are estimated at 4,000 birds from overseas and 2,500 from UK populations, so about 6,500 birds overall. These estimates also have a high uncertainty, especially regarding numbers from overseas populations which represent a major part of the totals.

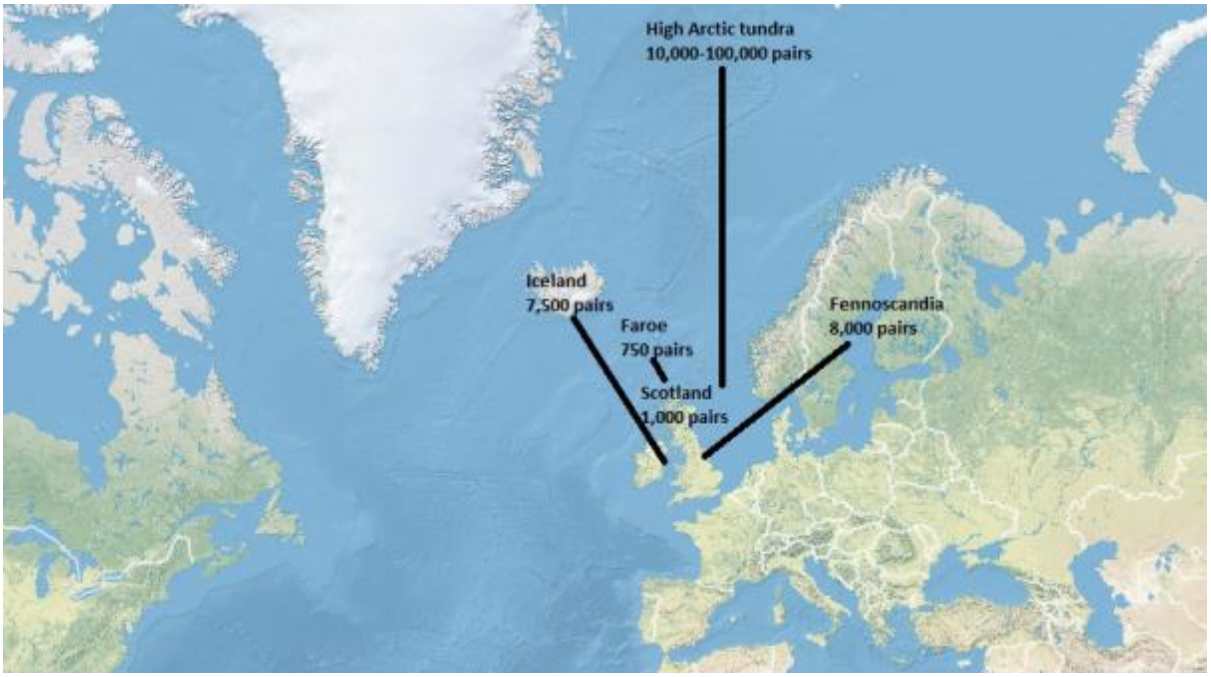


Figure 10.2. Breeding population origins of Arctic skuas in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

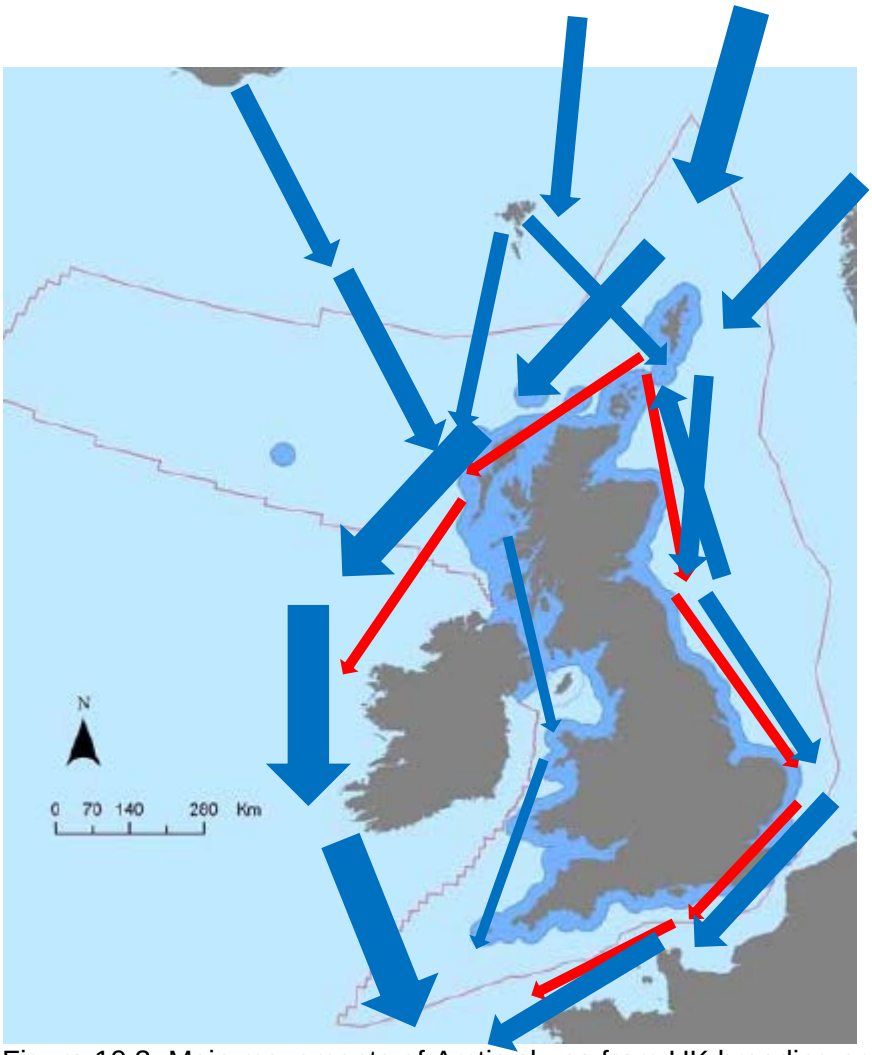


Figure 10.3. Main movements of Arctic skuas from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. See also Forrester et al. (2007) page 728.

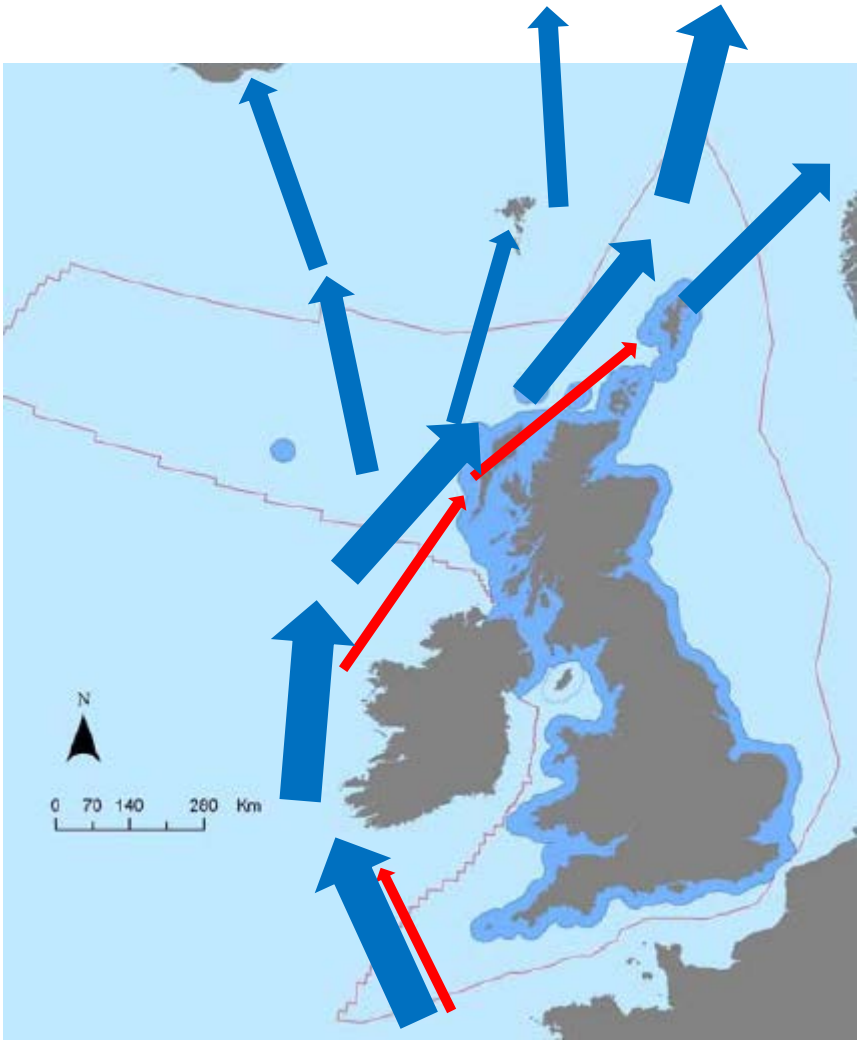


Figure 10.4. Main spring movements of Arctic skuas to UK breeding areas (red arrows) and towards overseas populations (blue arrows) through UK waters. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. See also Forrester et al. (2007) page 728.

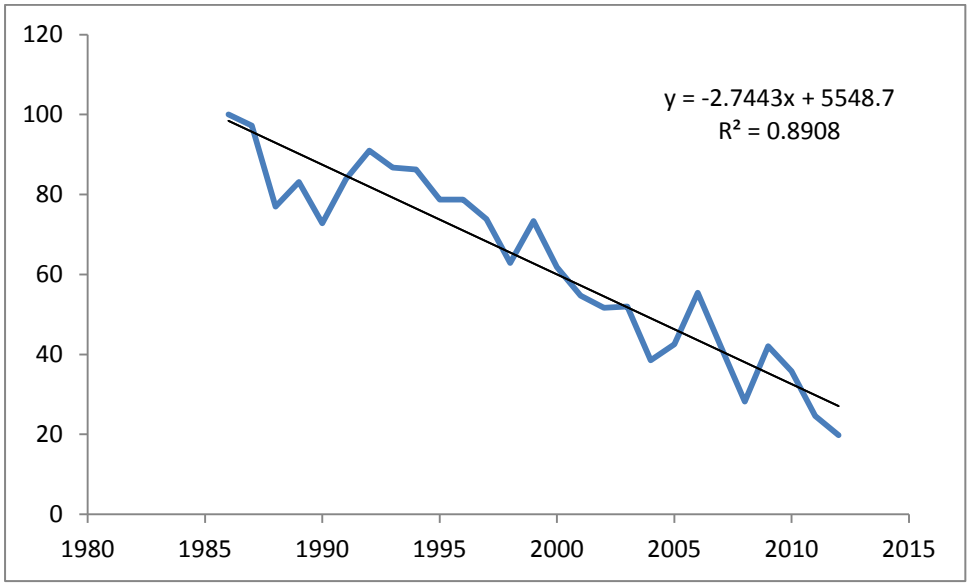


Figure 10.5. Trend in the Arctic skua breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

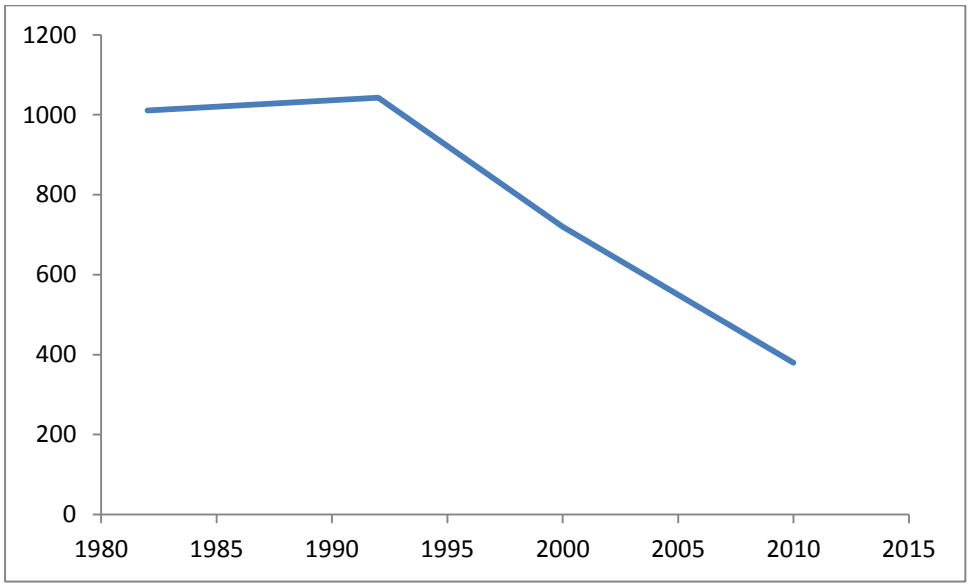


Figure 10.6. Trend in the Arctic skua breeding numbers in Orkney from 1982-2010. Data from Meek et al. (2011).

Data show a 31% decline in 8 years from 1992 to 2000, and a 47% decline in 10 years from 2000 to 2010 (Meek et al. 2011). Meek et al. (2011) concluded that declines in Arctic skua colonies in Orkney were related to colony size (a density-dependent relationship with larger colonies declining more than smaller ones) and to the numbers of great skuas in the area (an impact of predation, of mortality caused by fighting over territory ownership, and loss of nesting habitat to the larger species; see also Phillips et al. 1998).

10.9 Proportion of UK population from UK breeding SPAs

The 7 SPAs with breeding Arctic skuas as a feature together held 780 pairs at designation, estimated to represent ca. 24% of the British breeding population at that time (Stroud et al. 2001). Breeding numbers of Arctic skuas have declined very considerably since 2000 (Figures 10.5 and 10.6), with the decline being especially large at some of the largest colonies (which are the SPA populations). Therefore, the percent of the population breeding within the SPA suite for the species has decreased. Based on census data mostly from

around 2010, Stroud et al. (2014) estimated that the breeding Arctic skua SPA suite held 16.3% of the GB (=UK) population at that time. The sum of the most recent counts at each SPA is only 235 pairs (Table 10.1) whereas Stroud et al. (2014) summed counts dated mostly around 2010 to 343 pairs. So it is clear that the decline in the numbers at SPAs has continued, and so the percent of the UK population in the SPA suite for breeding Arctic skuas is likely to be less than the 16.3% estimated by Stroud et al. (2014). The exact percentage is difficult to assess because the total breeding population in the UK has not been surveyed recently, and numbers in areas where the species breeds at low density outwith SPAs may possibly not have declined as much. The percent in the SPA suite is therefore likely to now be around 15%, but might possibly be even lower than that as the large colony on Fetlar SPA has not been counted since 2002 when there were still 83 pairs there, and it is highly likely that numbers there are now much lower than that, given that other SPA populations in Shetland that were previously similar in numbers to Fetlar have fallen to only 30 or 40 pairs (Table 10.1).



Figure 10.7. The SPA suite for Arctic skua. These SPA populations are listed in Table 10.1.

Table 10.1. The UK SPA suite for breeding Arctic skuas.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
Fetlar	Shetland	130	1994	Recovering 2006	96 83	2001 2002	SMP database Stroud et al. 2014
Foula	Shetland	125	1995	Declined 2007	71 41 63 50 41 37 35	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database Lewis et al. 2012 SMP database Gear 2012 Gear 2013
Fair Isle	Shetland	74	1994	Maintained 2009	37 65 70 29 20 19	2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database FIBO Report SMP database
West Westray	Orkney	77	1996	Declined 2007	55 38 <27	2000 2007 2010	Stroud et al. 2014 Lewis et al. 2012 Meek et al. 2011
Papa Westray	Orkney	135	1996	Declined 2000	25 22	2011 2012	Lewis et al. 2012 Orkney Bird Report
Hoy	Orkney	59	2000	Maintained 2000	16 12	2010 2010	Meek et al. 2011 SCR database
Rousay	Orkney	180	2000	Declined 2007	114 46 37	2000 2007 2010	Lewis et al. 2012 Lewis et al. 2012 Meek et al. 2011

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

10.10 BDMPS

UK waters can be split into two BDMPS for Arctic skuas during migration seasons. The UK North Sea and Channel region holds about 6,000 birds in autumn and 1,000 in spring. The UK Western waters region holds about 5,000 birds in autumn and 5,000 in spring. These two areas should be treated as spatially separate BDMPS because although all breeding Arctic skua SPAs are in the UK North Sea and Channel area, much of the migration of this species passes through UK western waters. Therefore UK SPA birds are strongly represented in one BDMPS but not in the other. Details of apportioning of birds from different populations are given in Appendix A Tables 26 to 29. Since individual birds cannot be members of more than one spatially defined BDMPS, a minority of birds from colonies in the North Sea are (perhaps counter-intuitively) allocated to the UK western waters BDMPS rather than to the UK North Sea and Channel BDMPS. These are birds, predominantly from colonies in Shetland and Orkney, which migrate quickly out of, or into, the North Sea, but linger in UK western waters for some prolonged period during migration. These birds are therefore allocated to the BDMPS spatial area in which they spend more time, rather than necessarily being allocated into the BDMPS spatial area within which their breeding site happens to be located.

Based on evidence reviewed in sections 10.5, 10.6 and 10.7, it is estimated that in autumn 60% of adults and 40% of immatures from breeding Arctic skua UK SPA populations migrate through the UK North Sea and Channel waters, whereas 40% of adults and 30% of immatures migrate through UK western waters, whereas 100% of adults and 70% of

immatures from UK non-SPA western waters migrate through UK western waters. It is estimated that in autumn, 1% of adults and immatures from high Arctic populations migrate through UK North Sea and Channel waters and the same percentage through UK western waters, 2% of adults and immatures from Iceland migrate through UK North Sea and Channel waters and the same percentage through UK western waters, 10% of adults and immatures from Fennoscandia and Faroe migrate through UK North Sea and Channel waters, 5% of birds from Fennoscandia and 10% of birds from Faroe migrate through UK western waters (Appendix A Tables 26 and 27). This results in an estimate of 1,211 birds from UK and 5,216 from overseas in the autumn migration UK North Sea and Channel waters BDMPS and 1,439 birds from UK and 3,848 from overseas in the autumn migration UK western waters BDMPS.

Based on evidence reviewed in sections 10.5, 10.6 and 10.7, it is estimated that in spring, 40% of adults and 10% of immatures from breeding Arctic skua UK SPA populations migrate through the UK North Sea and Channel waters, whereas 60% of adults and 50% of immatures migrate through UK western waters, whereas 100% of adults and 70% of immatures from UK non-SPA western waters migrate through UK western waters. It is estimated that in spring, 0.2% of adults and 0.1% of immatures from high Arctic populations migrate through UK North Sea and Channel waters whereas 1% of birds from high Arctic populations migrate through UK western waters, 0.5% of adults and 0.1% of immatures from Iceland migrate through UK North Sea and Channel waters and 1% of Icelandic birds through UK western waters, 1% of adults and 0.5% of immatures from Fennoscandia migrate through UK North Sea and Channel waters, 5% of adults and 3% of immatures from Fennoscandia migrate through UK western waters, 0.5% of adults and 0.1% of immatures from Faroe migrate through UK North Sea and Channel waters and 5% of adults and 2% of immatures migrate through UK western waters (Appendix A Tables 28 and 29). This results in an estimate of 645 birds from UK and 582 from overseas in the spring migration UK North Sea and Channel waters BDMPS and 1,907 birds from UK and 3,204 from overseas in the spring migration UK western waters BDMPS.

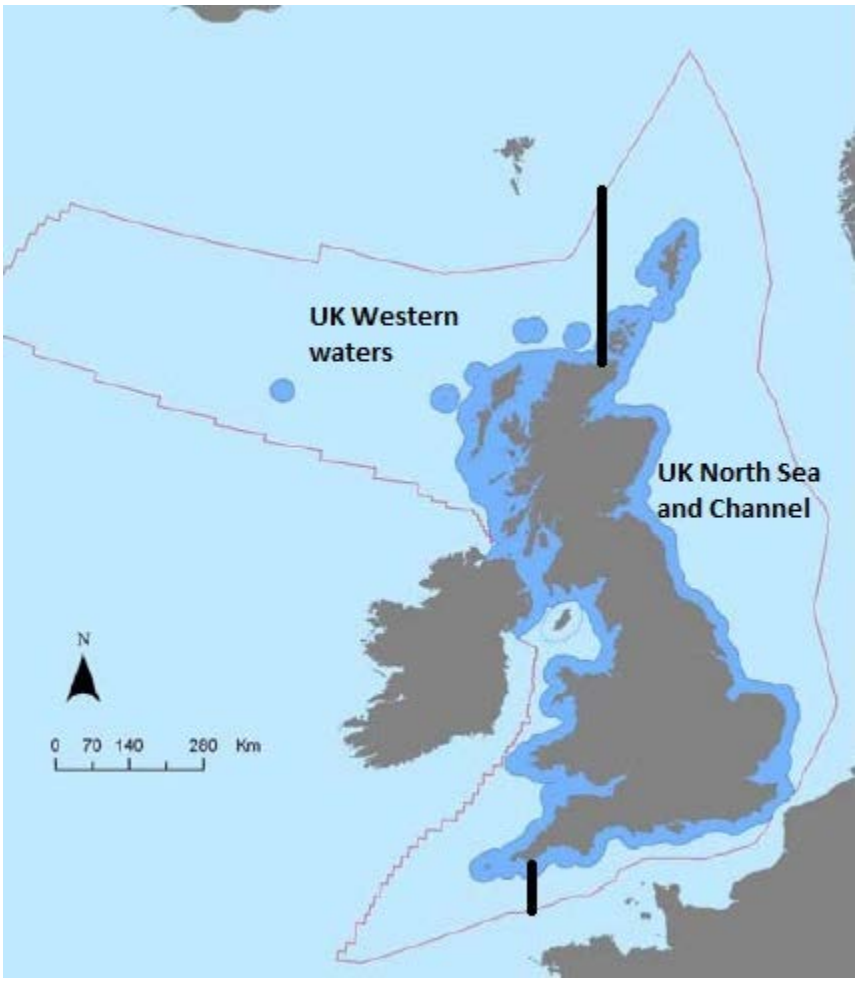


Figure 10.8. Two defined BDMPS spatial areas for Arctic skua: 'UK North Sea and Channel' and 'UK Western waters'.

10.11 Proportion of UK breeding SPA birds in BDMPS

During migration, the relatively small UK population (about 1,000 pairs, so 2,000 adults giving a total of about 3,000 birds of which many young immatures do not return from wintering areas to UK waters so a total of about 2,600 birds in UK waters) represents a minority of the birds present in UK waters. Probably UK birds represent about 20% of the birds present in UK waters on average during the migration months, but this percentage is very uncertain. The percentage is unlikely to be much higher than this, however, since most UK birds are dark phase, and the proportion of dark phase birds observed during migration watches at UK sites is generally small, indicating that a large majority of the birds originate from breeding areas further north where dark phase birds are at a frequency close to zero. Probably only about 15% of Arctic skuas from the UK colonies are from within the breeding Arctic skua SPA suite. However, since all the SPA populations and most of the species' breeding population in the UK, are in the NW North Sea area, and rather few Arctic skuas migrate northwards through that area, the proportion of SPA birds in that area in spring will be higher than in other BDMPSs. The proportion of the BDMPS represented by adults from UK SPA populations can be computed from data in Appendix A Tables 26 to 29. For example, in the UK North Sea and Channel BDMPS in autumn migration season there are 6,427 birds of which 281 are adults from UK SPA populations, so those represent 4.4% of the total present.

10.12 Spatial distribution of UK breeding SPA birds across the BDMPS

During autumn migration, birds dispersing from UK SPAs will all be in the North Sea and Channel BDMPS initially. However, these birds will move through this and some through the UK western waters BDMPS and often stop for some days in locations where there are opportunities to steal food from terns, so the distribution of SPA birds will quickly become fairly random across the BDMPSs. In spring, this process is likely to act in reverse, but with spring migration generally being somewhat faster and more direct towards colonies than in autumn.

11. GREAT SKUA *Stercorarius skua*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in autumn (August to October) (adults and immatures)	Numbers in UK waters in winter (November to February) (adults and immatures)	Numbers in UK waters in spring (March-April) (adults and immatures)
Overseas	30,000	5,562	1,363	5,655
UK	43,000	30,330	178	27,920
Total	73,000	35,892	1,541	33,575

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Autumn migration BDMPS (August to October)			
UK North Sea and Channel	19,556	2,141	17,415
UK Western waters	16,336	3,421	12,915
Winter BDMPS (November-February)			
UK North Sea and Channel	143	143	0
UK Western waters	1,398	1,220	178
Spring migration BDMPS (March-April)			
UK North Sea and Channel	8,485	982	7,503
UK Western waters	25,090	4,673	20,417

Colour coding for numbers of UK birds in the autumn migration BDMPS is amber. This reflects uncertainty about changes in breeding numbers at some UK colonies that have not been censused since Seabird2000. Breeding numbers have declined recently at some of the larger colonies but appear to still be increasing at some small colonies, and it is the latter that tend to lack recent census data. Colour coding for numbers of birds from overseas populations passing through UK waters in autumn is coded red because information on migrations of great skuas from Iceland, Faroe and Norway is based only on ring recovery data. Recoveries of pelagic or offshore seabirds tend to be highly biased because only a very small proportion of ringed birds are recovered, and many recoveries are associated with mortality related to human activities (such as fishery bycatch or birds being shot). There is only limited data from tracking birds equipped with geolocators (small numbers of breeding adults having been tracked from Iceland and Norway in only a single year). For these reasons, numbers in the winter BDMPS are coded red as are numbers in the spring migration BDMPS. The spring data are considered less reliable than the autumn data because spring passage results in very few ring recoveries, tends to occur over a shorter time period, and tends to occur in western waters which have lower survey coverage in the ESAS database than for North Sea waters and also have fewer and less consistently watched migration sites.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 30 to 35.

11.1 Breeding range and taxa

The species is monotypic (unless southern hemisphere taxa which do not visit European waters are included as conspecific which seems to be contrary to genetic evidence) and biometrics do not appear to help to identify origins of individuals. Great skuas breed in Scotland (9,634 pairs; Mitchell et al. 2004, but now decreased to probably about 8,900 pairs or less based on known declines at UK SPA colonies and assuming similar declines at other colonies), Faroe (500 pairs; Hammer et al. 2013), Iceland (5,400 pairs; Mitchell et al. 2004), Norway (360 pairs including Bear Island, Svalbard and Jan Mayen; Mitchell et al. 2004), and Russia (at least 10 pairs; Anker-Nilssen et al. 2000).

11.2 Non-breeding component of the population

Great skuas start to breed when 7 years old (BTO Birdfacts). Adult survival rate is 0.888 (BTO Birdfacts), juvenile survival 0.8 (BTO Birdfacts) and mean productivity is 0.664 chicks per pair (JNCC database, n=138 measurements). To obtain a stable population, survival of immatures was retained at 0.8 for juveniles, set at 0.82 for 1-year olds, 0.84 for 2-year olds, 0.86 for 3-year olds and 0.888 for older age classes. The model population comprised 41% adults, 14% juveniles and 45% older immatures. There are 1.42 immatures per adult.

11.3 Phenology

Breeding colonies in the UK are largely deserted by October, with modal departure in August (Pennington et al. 2004; Forrester et al. 2007). Autumn migration starts in August (Cramp et al. 1977-94; Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Peak autumn migration occurs in August-October in English waters (Brown and Grice 2005), early September (Pennington et al. 2004), September (Forrester et al. 2007), September-October (Cramp et al. 1977-94; Wernham et al. 2002), and July-October in Belgium (Vanermen et al. 2013). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in September and early October (Figure 11.1). Autumn migration is completed by late October (Forrester et al. 2007), early November (Pennington et al. 2004), November (Wernham et al. 2002) or early December (Cramp et al. 1977-94).

Spring migration starts in early March (Cramp et al. 1977-94; Pennington et al. 2004) or March (Wernham et al. 2002; Forrester et al. 2007). Peak spring migration occurs in January-April in Belgium (Vanermen et al. 2013) but the inclusion of January probably represents movement of very small numbers of birds, in March-April (Cramp et al. 1977-94), or in April in the UK (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late April (Figure 11.1). Spring migration is completed by May (Wernham et al. 2002), late May (Cramp et al. 1977-94; Pennington et al. 2004) or June (Forrester et al. 2007).

The first spring records of great skua in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were between 13 February and 24 April but mostly in late March, and the last records ranged from 11 October to 15 December but were predominantly in mid-November. Peak autumn migration was reported in August-September in most years, and peak spring migration was reported in April in most years. Birds re-occupy colonies from late March, with modal return in April (Pennington et al. 2004; Forrester et al. 2007).

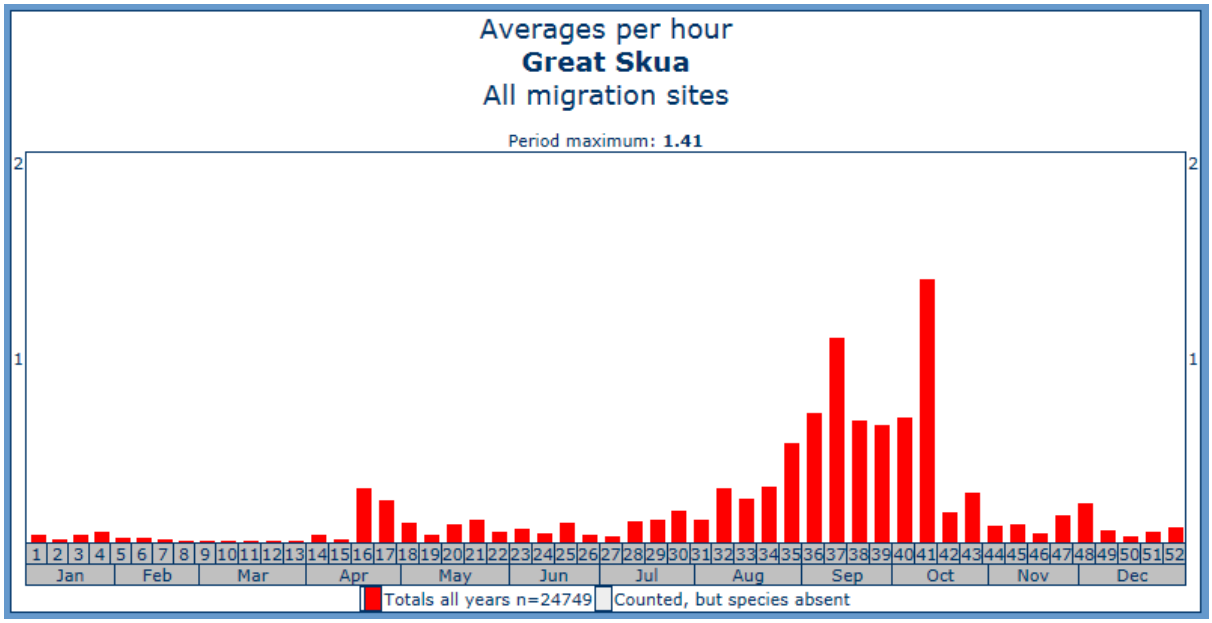


Figure 11.1. Average numbers of great skuas counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season September-April. From the data reviewed above, this would appear to be an appropriate definition.

11.4 Defined seasons:

- UK Breeding season — May-August
- Post-breeding migration in UK waters — August-October (**autumn BDMPS**)
- non-breeding season — September-April
- Return migration through UK waters — March-April (**spring BDMPS**)
- Migration-free breeding season — May-July
- Migration-free winter season — November-February (**winter BDMPS**)

Apart from the breeding season, three seasonal BDMPS periods are considered to be appropriate for great skua:

'Autumn' (post-breeding) migration BDMPS (August-October);

'Winter' BDMPS (November-February); and

'Spring' (pre-breeding) migration BDMPS (March-April).

11.5 Movements of birds from the UK population

Immatures (Klomp and Furness 1990) and failed breeders may leave colonies in July, followed in August-September by fledglings and successful breeders (Wernham et al. 2002; Furness 2010). Late breeders and young may not depart until October, and very small numbers remain in UK waters through to the end of the year and occasionally overwinter (Trektellen web site). Birds from UK colonies migrate over the Continental Shelf to the Bay of Biscay, Iberia or NW Africa. No adults from UK populations have been identified as wintering in North America. Only one or two ringed immatures from UK populations have been recovered on the coast of North America (Klomp and Furness 1992), so that region appears not to be visited by UK adults and not by significant numbers of UK immatures. Stable isotopes in feathers grown in the wintering area show location-specific signatures allowing individuals to be classified by major wintering areas: West Africa, southern Europe, or North America (Leat et al. 2013). Satellite tracking and deployment of geolocators on breeding great skuas suggests that numbers of adults wintering off west Africa may have increased, as numbers of ring recoveries from adult aged birds there were very small (Furness et al. 2006; Magnusdottir et al. 2012). Spring migration occurs in March-May, with rather rapid northwards movement mostly in April (Wernham et al. 2002; Trektellen web site). The high speed of spring migration may partly explain why there are far fewer ring recoveries in spring than in autumn (Wernham et al. 2002), but it also seems that most birds migrate northwards to the west of the British Isles with very few passing through the North Sea in spring, whereas during autumn migration much larger numbers are seen in the North Sea (Tasker et al. 1987; Forrester et al. 2007; Trektellen web site). As with most migrant seabirds, juveniles tend to winter further south, on average, than immatures which in turn tend to winter further south than breeding adults (Klomp and Furness 1992).

11.6 Movements of birds from overseas into UK waters

Migrants from all other breeding areas may pass through UK waters in autumn, at about the same time as UK birds are moving from colonies; there are autumn ring recoveries from birds ringed in Faroe and Iceland (Wernham et al. 2002). During autumn, peak numbers in the North Sea are seen in September (Tasker et al. 1987) and this pattern is also evident from seawatching data (Trektellen web site). While all breeders from UK colonies are thought to migrate through Europe to winter in southern Europe and off West Africa, about half of the breeders at colonies in Iceland and Bear Island migrate to winter off North America (Magnusdottir et al. 2012). A few of the birds wintering off North America also visit European waters during the same winter (Magnusdottir et al. 2012). Stable isotopes in feathers grown in the wintering area show location-specific signatures allowing individuals to be classified by major wintering areas: West Africa, southern Europe, or North America (Leat et al. 2013). Within the east Atlantic wintering range of the species, birds from Norway and Iceland tended to winter further north than those from UK (Magnusdottir et al. 2012). Great skuas from Faroe appear to show much the same migration and winter distribution as birds from UK colonies (Hammer et al. 2013). Thus, the very small numbers of great skuas present in UK waters in winter are more likely to be adults from Norway or Iceland than they are to be from UK colonies. Since the UK breeding numbers are twice those in Iceland, and numbers in Norway, Faroe and Russia are relatively small, birds from UK colonies predominate in the total population. In UK waters during migration, probably at least 80% of

birds are from UK colonies, since half of the birds from Iceland and Norway apparently travel to North America directly and do not pass through UK waters. In winter, however, the very small numbers of great skuas in UK waters may be predominantly adults from Iceland and Norway because those birds winter further north than birds from the UK.

11.7 Numbers in UK waters

Very few great skuas are present in English waters in winter, but small numbers are in the SW Approaches from November to March (Brown and Grice 2005). Very few (Forrester et al. 2007 estimate fewer than ten birds) are present in Scottish waters in winter. However, large numbers (relative to population size) migrate south through UK waters, especially through the North Sea, in autumn, and similar numbers migrate north through UK waters in spring, but predominantly to the west of the British Isles. Forrester et al. (2007) suggest that there are about 2,000 to 10,000 birds in Scottish waters in autumn, and about 1,000 to 6,000 in spring. These birds passing south inevitably also pass through English waters, as they winter off southern Europe or west Africa. It is reasonable to assume that almost the entire UK great skua population passes south through UK waters in autumn and all but the youngest age classes pass north through UK waters in spring (the youngest birds may remain in wintering areas all year, while middle ages of immature birds may migrate to Greenland and Norway in summer rather than stopping at UK breeding areas). The UK population is probably about 9,000 pairs at present, so 18,000 adults. Associated with this population are about 25,600 immatures, of which perhaps half will return to UK waters in summer and half be either in the wintering area or visit high latitudes rather than the UK in summer. So about 30,300 birds from the UK population are estimated to pass through UK waters on autumn migration. In addition, a few thousand birds from colonies in Norway, Russia, Faroe and Iceland pass through UK waters in autumn and spring. The exact number is not known, but the total is likely to be around 4,000 to 6,000 birds, as a large part of the Norwegian and Icelandic populations migrate west across the North Atlantic to Canadian waters, and some appear to migrate south from Iceland over the mid-Atlantic rather than via UK waters. These numbers are rather larger than the numbers suggested by Forrester et al. (2007) which presumably at least in part reflects the turnover that occurs with birds migrating through over a period of time, so that total numbers involved are larger than the ‘snapshot’ estimates provided by survey data.

11.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the species’ population, comprising 13,600 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 16,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 40,800 individuals. The biogeographic population with connectivity to UK waters is probably much the same as the total biogeographic population – so is estimated at 73,000 birds, 43,000 from the UK and 30,000 from overseas. This includes large numbers of immatures that do not necessarily return to UK waters but may range over areas from northern South America and west Africa to Greenland and the Barents Sea. Numbers in UK waters are estimated at 36,000 birds in autumn (August to October), 1,600 birds in winter (November to February), and 34,000 birds in spring (March and April).



Figure 11.2. Breeding population origins of great skuas in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

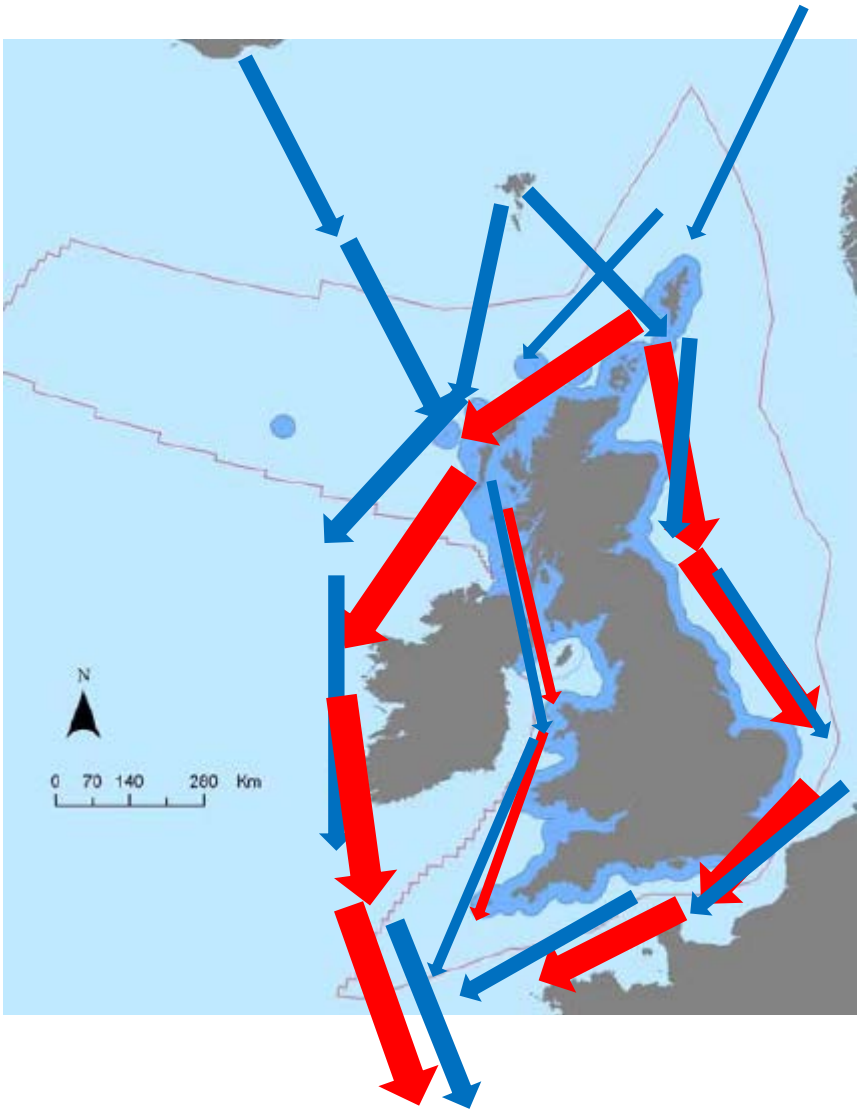


Figure 11.3. Main movements of great skuas from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes.

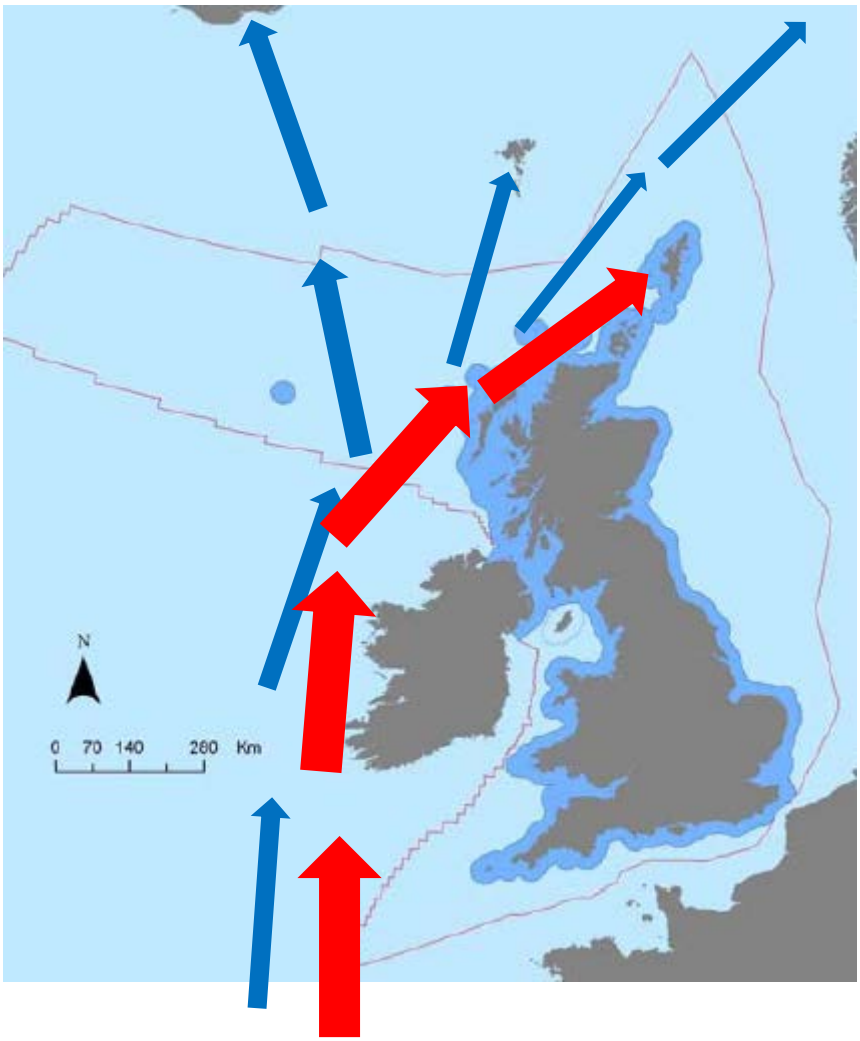


Figure 11.4. Main return movements of great skuas in spring to UK breeding areas (red arrows) and towards overseas populations (blue arrows) through UK waters. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes.

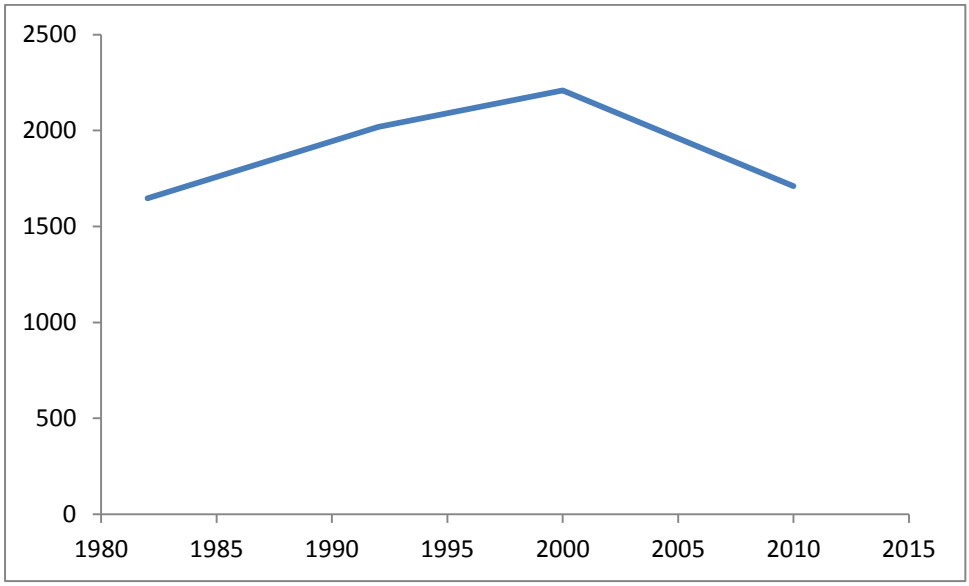


Figure 11.5. Trend in the great skua breeding numbers in Orkney from 1982-2010. Data from Meek et al. (2011).

Data show a 22.6% decline over the 10-year period between 2000 and 2010.

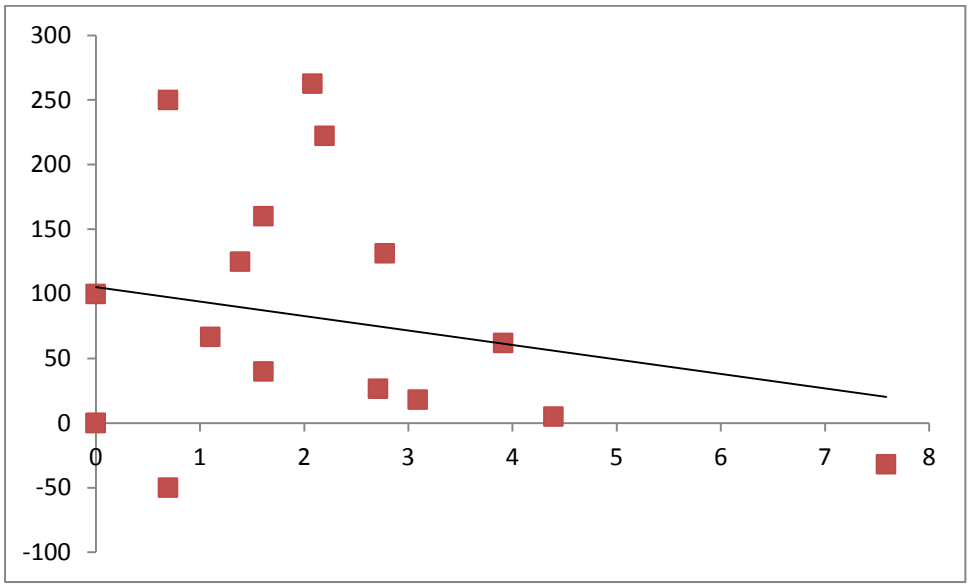


Figure 11.6. Rate of growth (% change in numbers) of breeding numbers of great skuas at colonies in Orkney between 2000 and 2010 in relation to size of the colony in 2000 (Natural Log). While the largest colony (Hoy) decreased considerably in numbers, many of the small colonies grew. Data from Meek et al. (2011).

11.9 Proportion of UK population from UK breeding SPAs

The 9 SPAs with breeding great skuas as a feature together held 6,262 pairs at designation, estimated to represent ca. 74% of the British breeding population (Stroud et al. 2001). Numbers have decreased since 2000 in Orkney, and at large SPA colonies in Shetland such as Foula, but have continued to increase at some smaller colonies. So the exact population size now is uncertain but is likely to be around 9,000 pairs. Because several of the largest colonies have decreased particularly markedly in size, and those are all SPA populations, the proportion of the UK population in the SPA suite for breeding great skuas will probably be less than it was previously. Based on data from years between 2000 and 2011, Stroud et al. (2014) estimated that 73.6% of the population was on SPAs. However, the figure may

now be closer to 70% due to continued large declines at Foula and Hoy in particular (the two largest colonies) and possibly some increases in areas that are not SPA populations where small numbers breed although those increases are very unlikely to be large enough to have much effect in reducing the overall decline in total breeding numbers that seems to be occurring (see Figure 11.8).

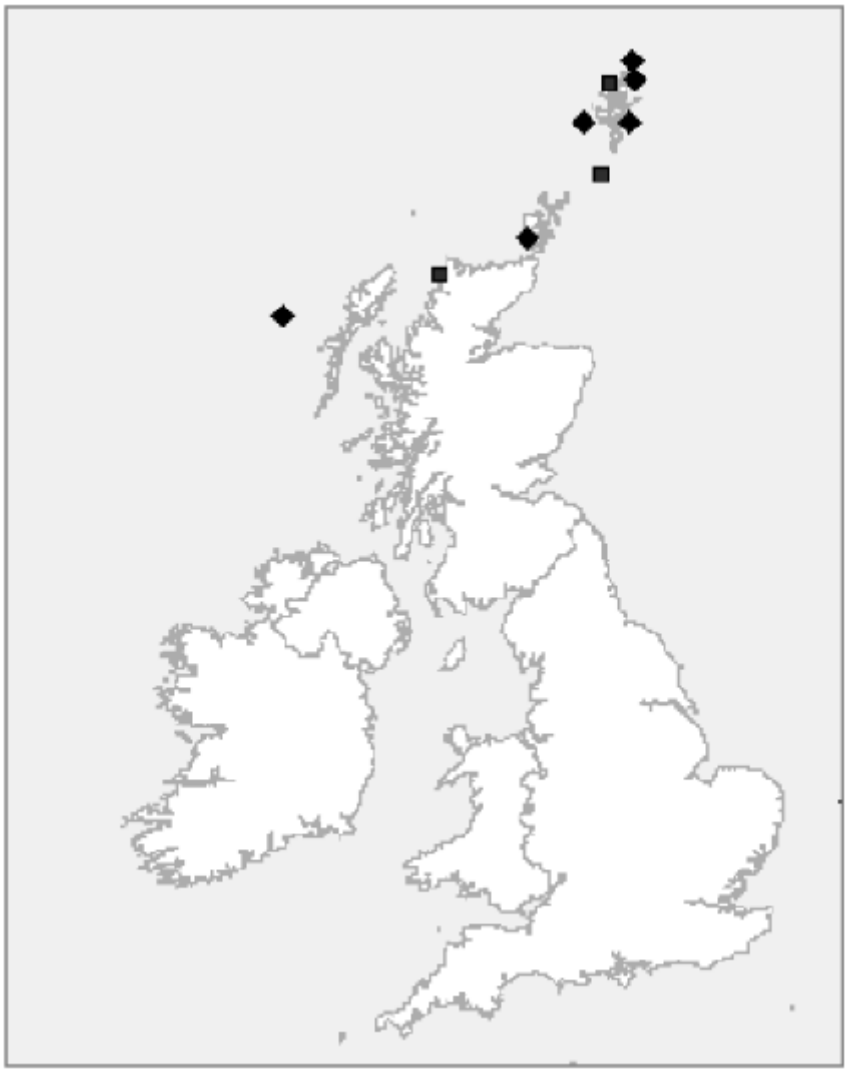


Figure 11.7. The SPA suite for great skua. These SPA populations are listed in Table 11.1.

Table 11.1. The UK SPA suite for breeding great skuas.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
Hermaness, Saxavord & Valla	Shetland	630	1994	Maintained 2013	726 751 979	2001 2007 2013	SMP database SMP database SMP database
Ronas Hill – North Roe & Tingon	Shetland	130	1997	Maintained 2002	189	2002	Stroud et al. 2014
Fetlar	Shetland	512	1994	Maintained 2006	593 585	2001 2002	SMP database Stroud et al. 2014
Foula	Shetland	2,170 (1992)	1995	Declined 2007	2,293 1,657	2000 2007	Seabird2000 SMP database

Noss	Shetland	410	1996	Maintained 2007	432 365 465	2001 2007 2013	Lewis et al. 2012 Lewis et al. 2012 SMP database
Fair Isle	Shetland	130	1994	Maintained 2009	280 227 300 266	2010 2011 2012 2013	SMP database SMP database FIBO Report SMP database
Hoy	Orkney	1,900 (1992)	2000	Maintained 2000	1,973 1,346	2000 2010	Seabird2000 Meek et al. 2011
UK Western waters							
Handa	NW Scotland	110	1990	Maintained 2000	212 202 190 272 266 241 135	2005 2006 2007 2008 2009 2010 2013	SMP database SMP database SMP database Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database
St Kilda	Western Isles	270 (1997)	1992	Maintained 2000	240 Hirta only: 210 189 139 174 151	2000 2000 2007 2008 2009 2012	Seabird2000 SMP database SMP database SMP database SMP database SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

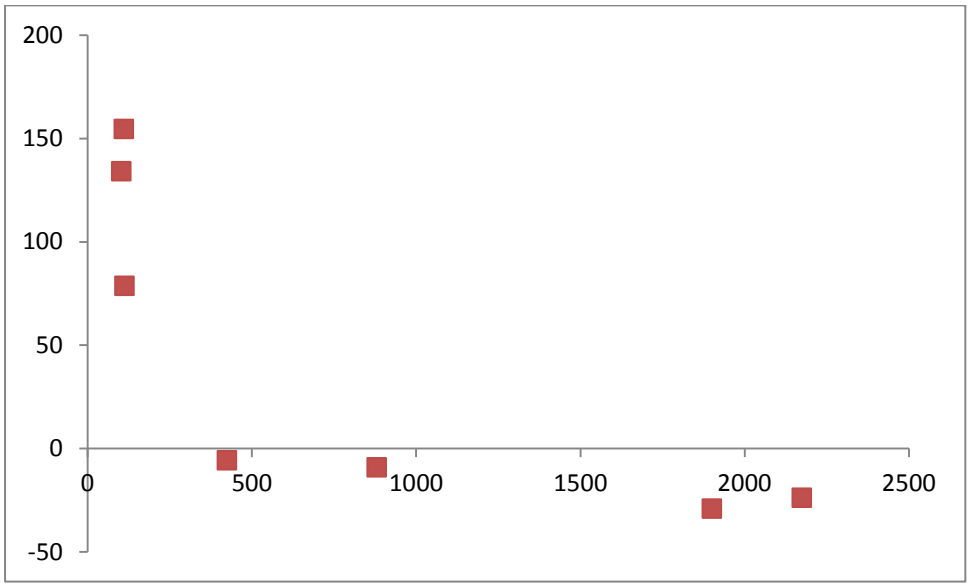


Figure 11.8. Percent change in numbers of pairs of great skuas from 1992 to 2010 at the largest colonies where count data are available (Foula, Hoy, Hermaness, Noss, St Kilda, Fair Isle, Handa). The data indicate that colonies of more than 400 pairs would decline in size while those with considerably fewer than 400 would grow. Data from Seabird Monitoring Programme database.

11.10 BDMPs

We need to consider three separate seasonal BDMPSs as the numbers in UK waters in winter are very much smaller than in autumn or spring, while in spring the migration route most used by great skuas is different from that used in autumn. We need to consider two spatial units for BDMPS; UK North Sea and Channel waters, and UK western waters. Most great skua colonies are in UK North Sea and Channel waters, but large numbers of migrants pass through UK western waters, especially in spring. Details of apportioning of birds into BDMPS are presented in Appendix A Tables 30 to 35.

Based on evidence reviewed in sections 11.5, 11.6 and 11.7, in autumn in the UK North Sea and Channel BDMPS, it is estimated that 60% of adults and 30% of immatures from colonies in the Northern Isles and Caithness will be members of the UK North Sea and Channel BDMPS, while 40% of adults and 20% of immatures will be members of the UK western waters BDMPS. This recognises that fact that a substantial number of birds from colonies in the northern isles move quickly during autumn migration into UK western waters but then spend some time there before moving further south to wintering areas, so those birds are allocated pro rata to the UK western waters BDMPS rather than to the UK North Sea and Channel BDMPS from which they departed from their breeding colonies at the end of the breeding season. No birds from colonies in the west of Scotland will be in the North Sea and Channel BDMPS whereas 100% of adults and 40% of immatures will be in the UK western waters BDMPS (Appendix A Tables 30 and 31). In addition, during autumn migration it is estimated that 10% of adults and 5% of immatures from Iceland, Norway and Faroe will be in the UK North Sea and Channel BDMPS, and 20% of adults and 5% of immatures from Iceland, 10% of adults and 5% of immatures from Norway, and 30% of adults and 5% of immatures from Faroe will be in the UK western waters BDMPS. These values result in an estimated BDMPS of 19,556 birds in the UK North Sea and Channel in autumn (17,415 originating from the UK), and 16,336 birds in the UK western waters BDMPS (12,915 originating from the UK).

Geolocator data loggers, satellite tracking data, and stable isotope analysis indicate that virtually all great skuas from the UK winter further south than UK waters with only a few adults wintering in the UK SW Approaches, whereas tracking data from adults nesting in Iceland and Norway show that birds from those populations tend to winter further north than birds from the UK. This implies that most, and apparently almost all, great skuas wintering in UK waters are birds from overseas populations. In the winter UK North Sea and Channel BDMPS there are thought to be no birds from UK colonies, and only very small numbers from overseas. Based on evidence reviewed in sections 11.5, 11.6 and 11.7, it is estimated that 1% of adults and 0.1% of immatures from Iceland, Norway and Faroe winter in UK North Sea and Channel waters (a total of 143 birds; Appendix A Table 32), while it is estimated that 1% of adults from UK colonies, 5% of adults and 0.1% of immatures from Faroe, 10% of adults and 0.1% of immatures from Iceland and Norway winter in UK western waters. This results in a BDMPS for UK western waters in winter of 1,398 birds. These totals appear to be reasonably consistent with evidence from the ESAS database and other at sea survey data which suggest a small winter hotspot for great skuas in the far SW of UK waters (Kober et al. 2010).

Based on evidence reviewed in sections 11.5, 11.6 and 11.7, spring migration of great skuas sees rather few birds moving north through the southern North Sea, but more pronounced migration through UK western waters, with many adults returning to colonies in the northern isles by way of western waters rather than through the North Sea. It is estimated that 30% of adults and 10% of immatures from UK North Sea colonies are in the UK North Sea and Channel spring BDMPS, whereas 70% of adults and 30% of immatures are in the UK western waters spring BDMPS (Appendix A Tables 34 and 35). 100% of adults and 40% of immatures from western colonies are in the UK western waters spring BDMPS. For birds

from overseas populations in spring, 5% of adults and 2% of immatures from Iceland, Norway and Faroe are estimated to be in the UK North Sea and Channel spring BDMPS, whereas 30% of adults and 5% of immatures from Iceland, 20% of adults and 5% of immatures from Norway, and 40% of adults and 5% of immatures from Faroe are in the UK western waters spring BDMPS (Appendix A Tables 34 and 35). This gives estimated BDMPSs for spring of 8,485 birds in the UK North Sea and Channel, and 25,090 birds in the UK western waters.

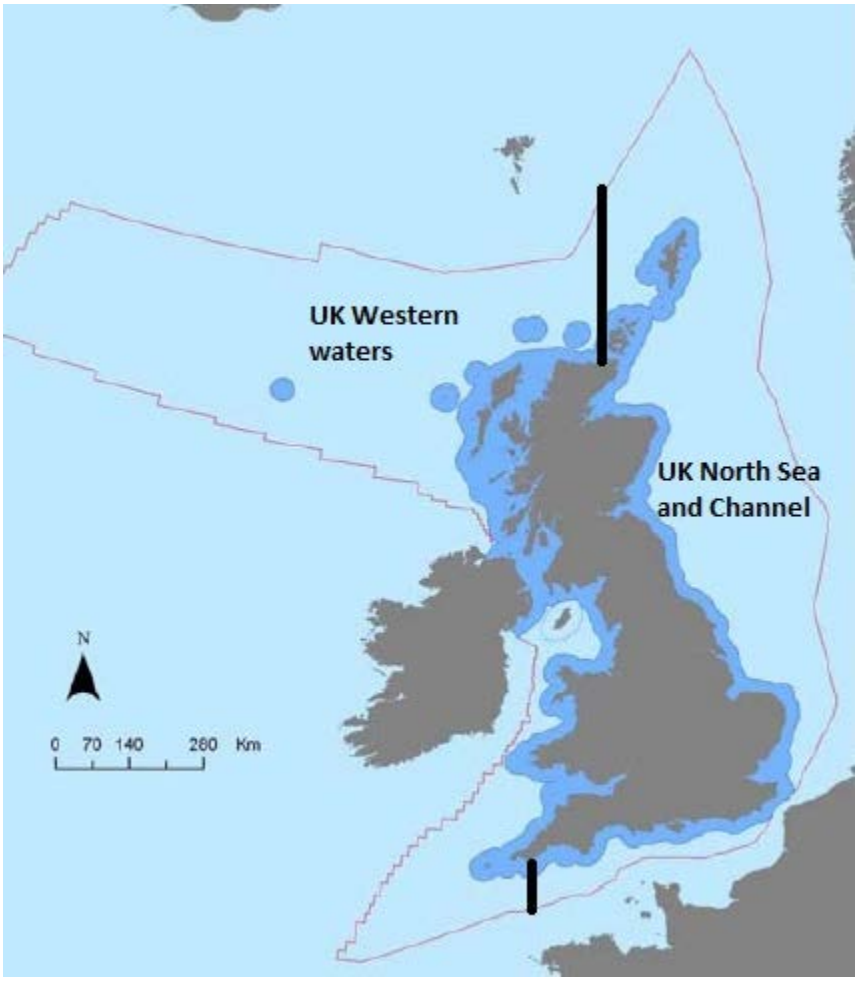


Figure 11.9. Two defined BDMPS spatial areas for great skua: 'UK North Sea and Channel' and 'UK Western waters'.

11.11 Proportions of UK SPA birds in each BDMPS

The UK suite for breeding great skuas is very strongly concentrated in the NW North Sea, with only small numbers in the West of Scotland region (Handa 135 pairs, St Kilda 151 pairs). The birds from SPA populations in the NW North Sea do not all migrate south through the North Sea; a proportion migrate southwards via the west of the British Isles. So the proportions of UK SPA birds in the different BDMPS in autumn and spring are not dramatically different despite the concentration of SPA birds being in Orkney and Shetland. Proportions can be computed from data in Appendix A Tables 30, 31, 34 and 35. For example, 6,584 adults from great skua breeding UK SPAs are in the UK North Sea and Channel autumn BDMPS which totals 19,556 birds, so adults from SPA colonies represent 34% of the total present. In UK western waters in autumn, adults from SPA colonies total 5,022 birds out of a population of 16,336, or 31%. Wintering birds in each BDMPS are likely to be predominantly from colonies in Norway and possibly Iceland, as those birds winter further north, on average, than birds from the UK. Data in Appendix A Tables 32 and 33 can

be used to estimate the proportion of each winter BDMPS comprising adults from breeding great skua UK SPAs. In the winter UK North Sea and Channel BDMPS this proportion is 0% adults from UK SPA colonies. In the winter UK western waters BDMPS there are estimated to be 116 adults from breeding great skua UK SPAs, from a BDMPS of 1,398 birds, so about 8% are adults from UK SPAs.

11.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Great skuas disperse from colonies in all directions at the end of the breeding season, and so the SPA birds will be mixed with non-SPA birds across the BDMPS. Aggregations of SPA birds are unlikely except to the extent that in Shetland some adults may attend colonies late into autumn, so there is likely to be some tendency for proportions of SPA birds to be locally higher close to the main SPA sites into the autumn, and birds returning early in spring may similarly aggregate in waters close to colonies before returning to their breeding territories onshore. However, aggregations are not likely to be pronounced, and there will be considerable mixing of birds from different populations.

12. LESSER BLACK-BACKED GULL *Larus fuscus*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in autumn (August to October) (adults and immatures)	Numbers in UK waters in winter (November to February) (adults and immatures)	Numbers in UK waters in spring (March-April) (adults and immatures)
Overseas	572,000	105,969	15,350	94,445
UK	292,000	266,342	65,123	266,342
Total	864,000	372,311	80,473	360,787

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Autumn migration BDMPS (August-October)			
UK North Sea and Channel	209,007	62,870	146,137
UK Western waters	163,304	43,099	120,205
Winter BDMPS (November-February)			
UK North Sea and Channel	39,314	7,724	31,590
UK Western waters	41,159	7,626	33,533
Spring migration BDMPS (March-April)			
UK North Sea and Channel	197,483	51,346	146,137
UK Western waters	163,304	43,099	120,205

Numbers of lesser black-backed gulls in colonies in the UK are moderately well documented, with most SPA populations counted in at least one year since completion of Seabird2000. Moderate but fairly consistent declines in breeding numbers since 2000 are indicated both

by the JNCC seabird monitoring data and by examination of SPA colony counts. Thus data on numbers of UK lesser black-backed gulls migrasting through UK waters are coded amber. However, numbers of overseas lesser black-backed gulls passing through UK waters on migration are less well known. Information is mainly from ring recovery data (but including very extensive and detailed colour ringing studies from the Netherlands). Populations of lesser black-backed gulls overseas are large, and although only small or very small proportions of these birds migrate through UK waters, this increases the uncertainty about numbers passing through UK waters so estimated numbers of overseas birds are coded red. Numbers of lesser black-backed gulls wintering in UK waters seem to vary from year to year, presumably in relation to weather or food abundance. These numbers have increased over recent decades, but there is further uncertainty regarding the extent to which these birds spend time at sea or in terrestrial habitats. Wintering numbers in BDMPS are coded red both for numbers from overseas and from UK.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 36 to 41.

12.1 Breeding range and taxa

Three subspecies of lesser black-backed gull breed in Europe, but biometrics of individuals do not seem to have been used to identify origins of individuals. The subspecies *fuscus* breeds in Finland, northern Norway and northern and eastern Sweden, and has a distinct migration pattern, moving to winter in east Africa (Bustnes et al. 2013). Birds from that subspecies (which are relatively easy to identify in the field from plumage features) only occur in UK waters as vagrants. The subspecies *graellsii* breeds in Iceland, Faroe, the British Isles, and western Europe south to Portugal, and winters predominantly in Iberia or on the coast of northwest Africa. The subspecies *intermedius* breeds in Denmark, southern Norway and southern Sweden while populations somewhat intermediate between *intermedius* and *graellsii* breed in Germany and the Netherlands (Wernham et al. 2002). Birds from populations of *intermedius* show much the same migration patterns as birds from *graellsii* (Wernham et al. 2002).

12.2 Non-breeding component of the population

Lesser black-backed gulls start to breed when 4 years old (BTO Birdfacts). Adult survival rate is given as 0.913 in BTO Birdfacts (but more recent work on this species indicates a decline in survival with time for the population at Skomer http://jncc.defra.gov.uk/page-2886 so a lower value could be used but would have only a small influence on the ratio estimate because of corresponding adjustment of immature survival rates in the opposite direction to achieve a stable population trend), juvenile survival unknown (BTO Birdfacts) and mean productivity is 0.517 chicks per pair (JNCC database, n=66 measurements). To obtain a stable population, survival of immatures was adjusted to 0.7 for juveniles, 0.74 for 1-year olds, 0.79 for 2-year olds, 0.84 for 3-year olds. The model population comprised 60% adults, 15% juveniles and 25% older immatures. There are 0.68 immatures per adult.

12.3 Phenology

Breeding colonies in the UK are deserted by September, with modal departure in late July or early August (Pennington et al. 2004; Forrester et al. 2007). Autumn migration starts in late June (Pennington et al. 2004), July (Cramp et al. 1977-94; Forrester et al. 2007) or mid-July (Wernham et al. 2002). Peak autumn migration occurs in August (Pennington et al. 2004), August-September (Wernham et al. 2002), September (Forrester et al. 2007), and June-October in Belgium (Vanermen et al. 2013) and August-November throughout Europe and North Africa (Cramp et al. 1977-94). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred rather consistently through August-November (Figure 12.1) suggesting a very protracted autumn

migration through UK waters. Autumn migration is completed by early October in Shetland (Pennington et al. 2004) but not until October-November (Wernham et al. 2002) or November (Forrester et al. 2007) or early December (Cramp et al. 1977-94) in the UK as a whole.

Spring migration starts in February in the winter quarters (Cramp et al. 1977-94), mid-February (Wernham et al. 2002) or late February (Forrester et al. 2007) in the UK as a whole, or early March (Pennington et al. 2004) in Shetland. Peak spring migration occurs in February-April in Belgium (Vanermen et al. 2013), in March (Wernham et al. 2002; Forrester et al. 2007), March-April (Cramp et al. 1977-94), or in April in Shetland (Pennington et al. 2004). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in early March, although there were suggestions of a further peak in mid-April (Figure 12.1). Spring migration is completed by April (Wernham et al. 2002) or May (Cramp et al. 1977-94; Forrester et al. 2007) or early June in Shetland (Pennington et al. 2004).

The first spring records of lesser black-backed gull in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 2 January to 1 April, but mostly in February, and the last records were from 25 August to 29 December, but mostly in late October. Peak autumn dispersal/migration was reported in July-August in most years, and peak spring migration was reported in March in most years. Birds re-occupy colonies from late February or early March with modal return in late March (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007).

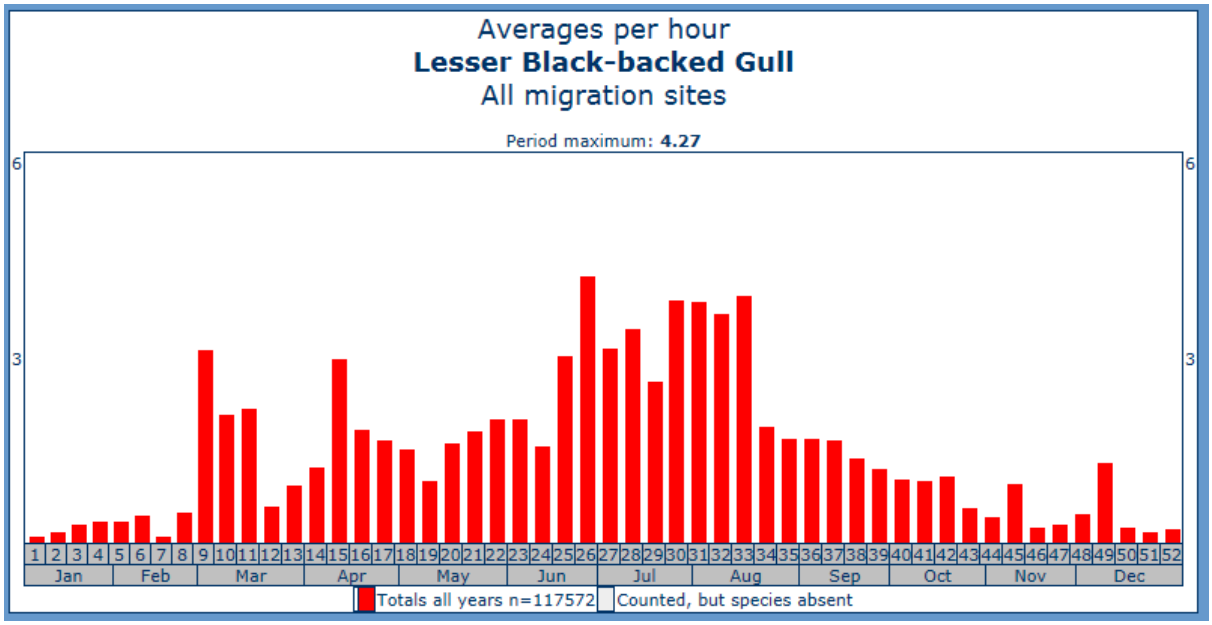


Figure 12.1. Average numbers of lesser black-backed gulls counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season September-April. However, from the data reviewed above, a more appropriate definition would be breeding season April-August, non-breeding season September-March.

12.4 Defined seasons:

- UK Breeding season — April-August
- Post-breeding migration in UK waters — August-October (**autumn BDMPS**)
- non-breeding season — September-March
- Return migration through UK waters — March-April (**spring BDMPS**)
- Migration-free breeding season — May-July
- Migration-free winter season — November-February (**winter BDMPS**)

Apart from the breeding season, three seasonal BDMPS periods are considered to be appropriate for lesser black-backed gull:

'Autumn' (post-breeding) migration BDMPS (August-October);

'Winter' BDMPS (November-February); and

'Spring' (pre-breeding) migration BDMPS (March-April).

12.5 Movements of birds from the UK population

In the UK, autumn movements start in the second half of July. Migration southwards is fairly rapid from northern colonies, with most birds away by August (Orkney Bird Reports; Shetland Bird Reports), but is protracted in southern Britain where some birds remain near colonies until early October (Wernham et al. 2002). Timing of dispersal from colonies is the same in The Netherlands; occurring in July-August (Camphuysen 2013). Many fledglings are accompanied by their parents during initial autumn dispersal, but it is unclear if families remain together during autumn migration. Camphuysen (2013) found that successful breeders abandoned the colony when their young were about 50 days old, and that southward autumn movement started first in immatures, then in adults, and last in juveniles, suggesting that post-fledging care of juveniles was mostly minimal. Camphuysen (2013) reported that movement away from colonies in The Netherlands occurred earlier in autumn in years since 2000 than it had previously, suggesting deteriorating conditions in the breeding areas. Young birds tend to move further south than adults (Wernham et al. 2002). Some adults apparently tend to return each year to the same wintering site, although some may change wintering areas between years. Adults return to colonies in the UK in February to April (Wernham et al. 2002), with some evidence for birds that winter furthest north arriving back at colonies first. Until the 1950s the lesser black-backed gull in the UK was considered to be a migrant, with all birds wintering in southern Europe or north Africa. However, in the 1960s and 1970s increasing numbers, mostly of adults, remained in the UK overwinter (Wernham et al. 2002). This change may relate as much to availability of land-fill feeding sites as to warming of the climate (Banks et al. 2007). There were estimated to be about 70,000 lesser black-backed gulls wintering in Britain and Ireland in censuses held in 1985 and 1993 (Wernham et al. 2002), and 125,113 in 2003-06 (Burton et al. 2013) suggesting that numbers have continued to increase. Not only did winter distribution change, but migration routes also changed, with increasing numbers migrating overland. Recent tracking studies by the British Trust for Ornithology of breeding adults from a colony in East Anglia found that although autumn migration was predominantly coastal, the more rapid spring migration from north Africa to England occurred overland through central France.

12.6 Movements of birds from overseas into UK waters

Foreign-ringed lesser black-backed gulls recovered in Britain and Ireland have come from Iceland, Faroe, Norway, Sweden, Finland, Denmark, the Netherlands, Belgium, the Channel Islands and Spain; almost 60% of these are likely to be from the subspecies *intermedius* mostly from breeding sites in Norway, Sweden and Denmark, while the remaining 40% are

predominantly *graellsii* from Iceland and Faroe (Wernham et al. 2002). The single recovery of a bird of the subspecies *fuscus* from Finland can be discounted as exceptional, as that subspecies can be identified in the field from plumage features, and is only very rarely seen in the UK (Wernham et al. 2002). Most foreign-ringed lesser black-backed gulls from Norway, Sweden, Denmark, the Netherlands and Belgium have been found in SE England (Wernham et al. 2002), suggesting that these continental birds cross the southern North Sea. Birds from Iceland and Faroe have been more broadly distributed through the British Isles. However, lesser black-backed gulls from colonies in The Netherlands mostly winter in France, Portugal and Spain, and relatively few birds marked in The Netherlands have been seen in the UK (Camphuysen 2013), although there are a few records. Seabird 2000 reported 87,413 pairs in UK, 3,800 pairs in Ireland, 25,000 pairs in Iceland, 9,000 pairs in Faroe, 25,000-36,000 pairs in Norway, 15,000-20,000 pairs in Sweden (however BirdLife International (2004) cite 2000-5000 pairs in Sweden but without listing the data source), 4,400 pairs in Denmark, 32,000-57,000 pairs in the Netherlands. In the Netherlands, breeding numbers peaked around 2005 (Camphuysen 2013) at around 90,000 pairs and are probably now around 80,000 pairs (Camphuysen 2013).

12.7 Numbers in UK waters

Musgrove et al. (2013) report that there are 120,000 in Britain in winter, 130,000 in UK in winter, but it is unclear if these include birds at sea as well as onshore and at coastal roosts. From surveys in 2007 and 2008, Fauchald and Tveraa (2009) reported mean densities at sea of 0.7-10 birds per km^2 in the Norwegian Sea in spring/summer, and 0 birds per km^2 in the Barents Sea in autumn. Lesser black-backed gulls are distributed throughout the North Sea in summer but with much higher densities in the southeastern North Sea and low densities in the northwestern North Sea (Skov et al. 1995; Camphuysen 2013). About 130,000 birds were estimated to be in the North Sea in March-August (Skov et al. 1995) (although this estimate was based on data that are now rather out of date), with about 95% of these in the eastern half of the North Sea (Camphuysen 2013). Areas of greatest importance for this species in the North Sea are between Vlieland and Ijmuiden (off Texel) from May to October, in the Skagerrak in March-April and Helgoland Bight in May-June (Camphuysen 2013). Lesser black-backed gulls show a strong association with the distribution of fishing vessels in the southern North Sea in summer, congregating in areas where fisheries discards are available (Camphuysen et al. 1995), so their distribution reflects the locations of large colonies and also the behaviour of fisheries in the area. In winter, the North Sea is largely abandoned, but about 15,000 birds spend the winter in the English Channel (Camphuysen 2013). According to Brown and Grice (2005) highest numbers in English waters in winter are found in the Celtic and Irish Seas and SW Approaches. Wintering numbers inland in England have increased from 165 in 1953 to 6,960 in 1963, 15,823 in 1973, 36,154 in 1983, and 27,230 in 1993 (Brown and Grice 2005). It is estimated that there were 70,000 lesser black-backed gulls wintering in England (inland plus English waters) in the 1980s, and that numbers have increased since then (Brown and Grice 2005). Forrester et al. (2007) suggest that only about 200-600 birds winter in Scotland but that there are 30,000-50,000 in spring passage and 50,000-80,000 in autumn passage. Bradbury et al. (in press) used ESAS and offshore wind farm survey data to compare the relative importance of different marine areas at different times of year.

12.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the subspecies *graellsii* population, comprising 124,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 179,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 550,000 individuals. The biogeographic population with connectivity to UK waters totals about 292,000 birds (adults plus immatures) from the UK plus 572,000 birds (adults plus immatures) from overseas populations (Iceland, Norway, Sweden, Denmark, Faroe, Ireland, and The Netherlands). However, only small proportions

of the birds from overseas populations visit UK waters, so the estimated total numbers in UK waters are much smaller than this total. In autumn (August to October) there are estimated to be 372,000 birds in UK waters, 266,000 from UK and 106,000 from overseas. In winter (November to February) there are estimated to be 80,000 birds in UK waters, 65,000 from the UK and 15,000 from overseas. In spring (March and April) there are estimated to be 360,000 birds in UK waters, 266,000 from UK and 94,000 from overseas.

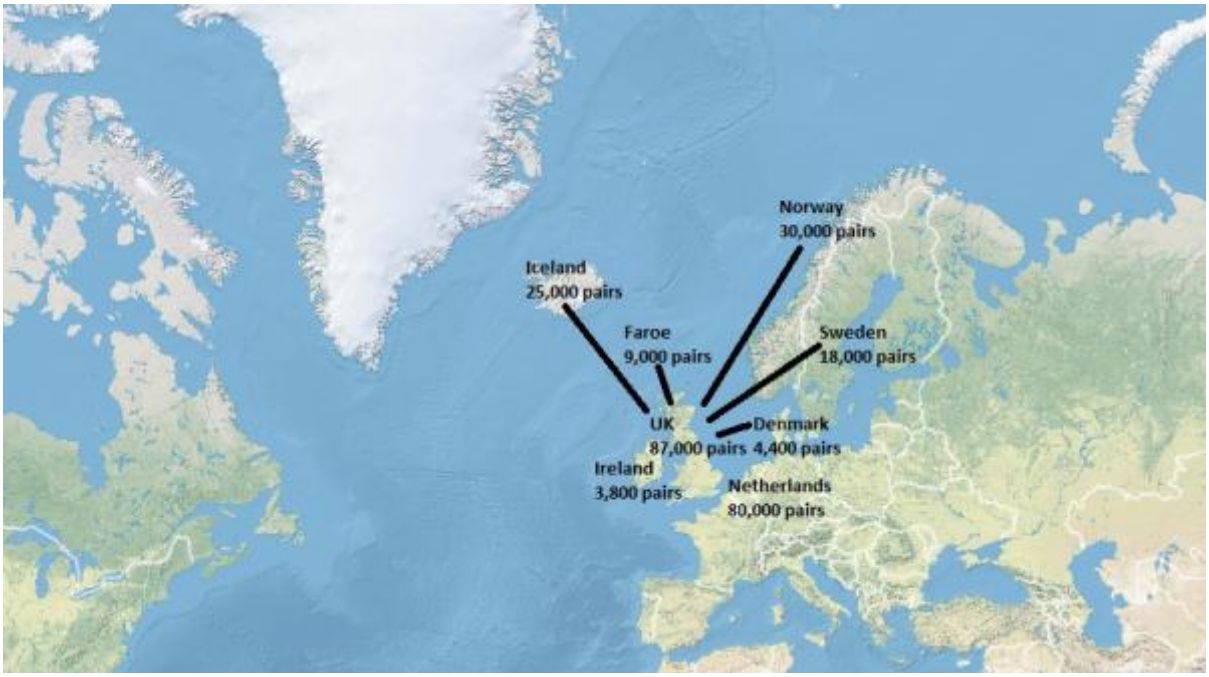


Figure 12.2. Breeding population origins of lesser black-backed gulls in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

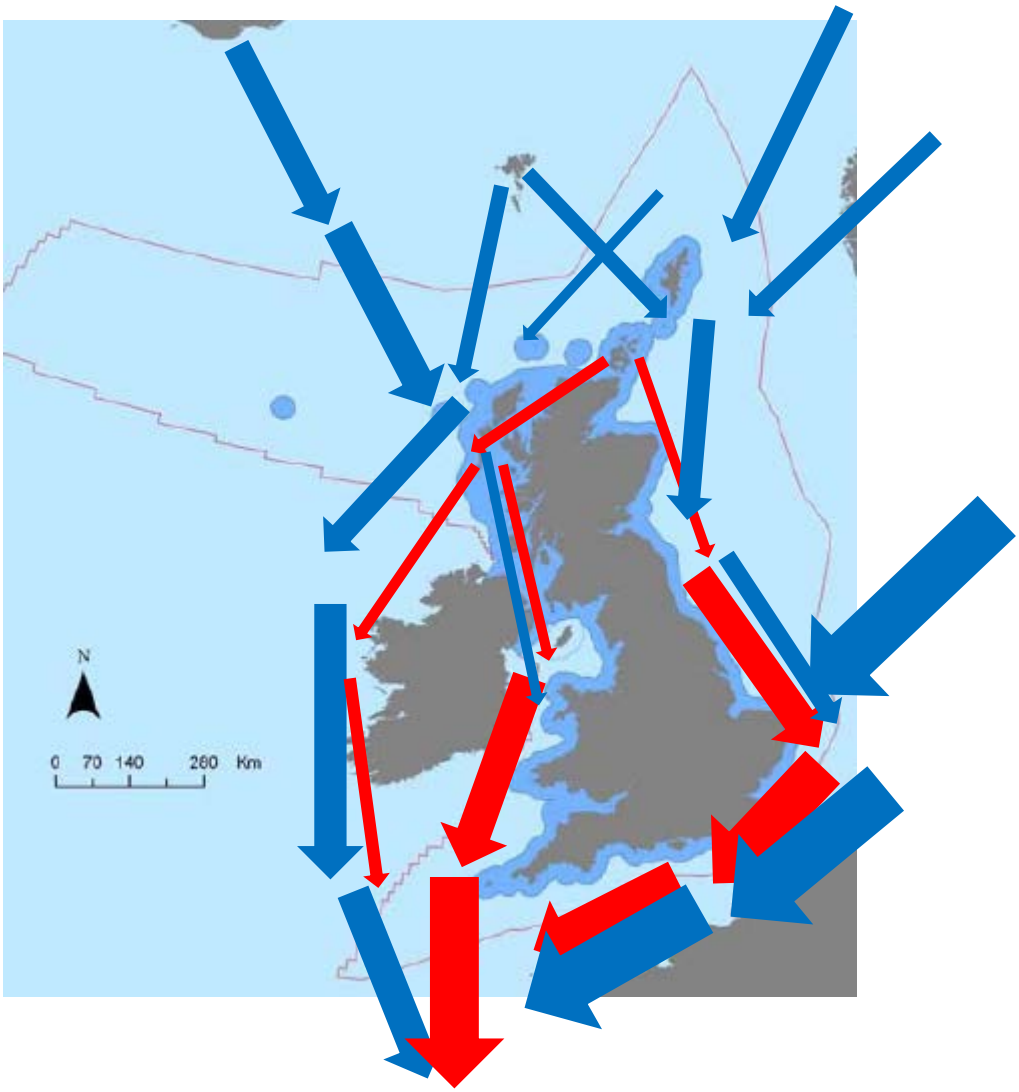


Figure 12.3. Main movements of lesser black-backed gulls from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes.

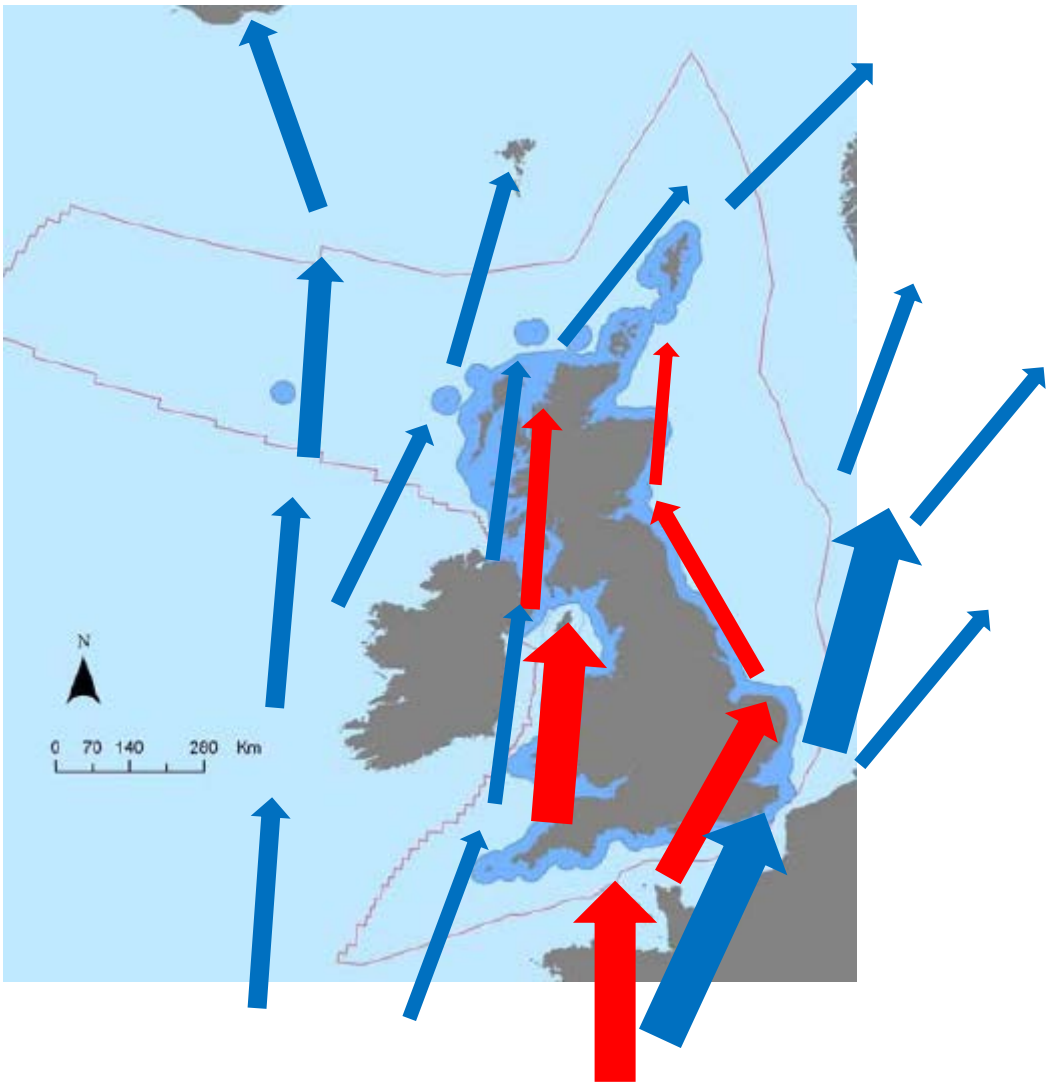


Figure 12.4. Main return movements of lesser black-backed gulls in spring to UK breeding areas (red arrows) and towards overseas populations (blue arrows) through UK waters. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes.

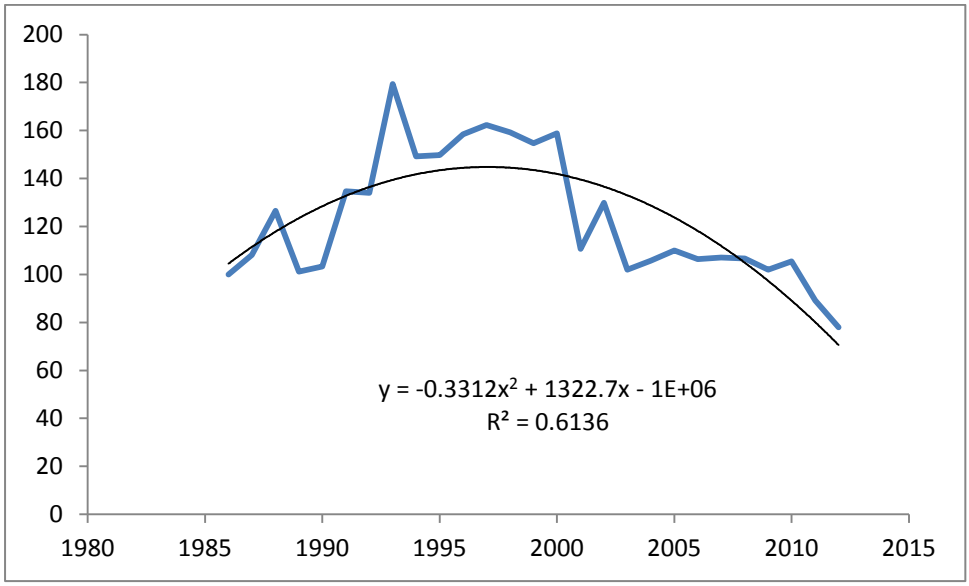


Figure 12.5. Trend in the lesser black-backed gull breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

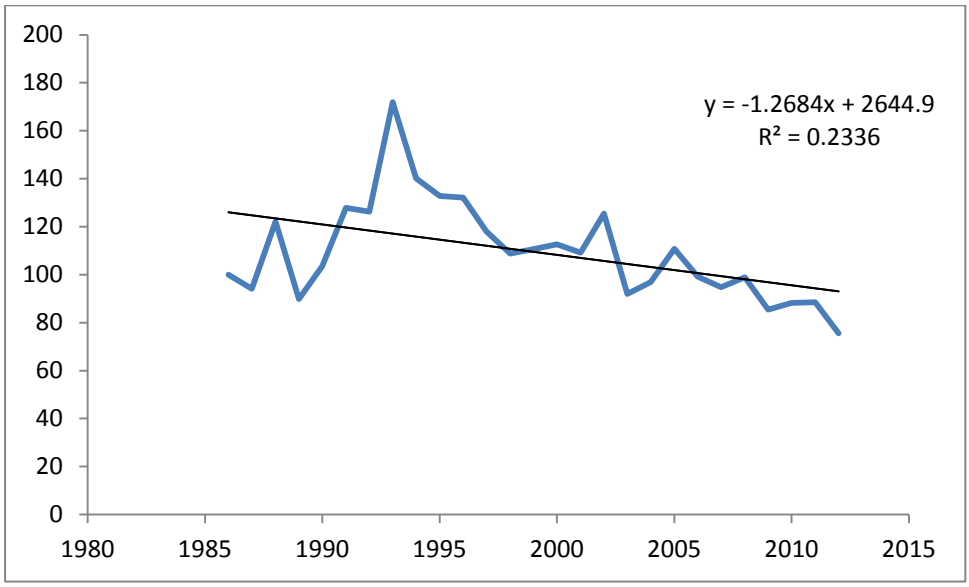


Figure 12.6. Trend in the lesser black-backed gull breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

12.9 Proportion of UK population from UK breeding SPAs

The 10 SPAs with breeding lesser black-backed gulls as a feature together held 88,633 pairs at designation, estimated to represent ca. 100% of the British breeding population (Stroud et al. 2001). However, this clearly overestimates the proportion on SPAs as there have been non-SPA colonies with substantial numbers for many decades. The 2014 UK SPA review (Stroud et al. 2014) reported that the UK breeding SPA populations represented 38.5% of the GB population in 2003-11, this large decrease being due to very large declines in breeding numbers at some of the largest colonies (all of which are SPAs).



Figure 12.7. The SPA suite for lesser black-backed gull. These SPA populations are listed in Table 12.1.

Table 12.1. The UK SPA suite for breeding lesser black-backed gulls.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
Forth Islands	E Scotland	1,500 (1985) Or 2,920 (Stroud et al. 2001)	1990	Maintained 2008	2,013 >2,100 1,608	2002 2008 2005-2009	Lewis et al. 2012 Lewis et al. 2012 Stroud et al. 2014
Alde-Ore Estuary	SE England	14,070 (1994-1998) Or 21,700 (Stroud et al. 2001)	1996	Counts may relate to just Orfordness and may exclude Havergate Marshes; there were 1747 AON there in 2013	6,000 5,000 1,678 1,584 900 550 550 640	2003 2006 2007 2008 2009 2010 2011 2012	Stroud et al. 2014 SMP database SMP database SMP database SMP database SMP database SMP database SMP database

UK Western waters							
Ailsa Craig	W Scotland	1,800 (1987)	1990	Declined 2010	183	2010	Lewis et al. 2012
Rathlin Island	N Ireland	155 (1985)	1999		127 36 107	1999 2007 2011	SMP database SMP database Stroud et al. 2014
Lough Neagh & Lough Beg	N Ireland	450 (Stroud et al. 2001)	1996		385 493	2000 2000	SMP database Stroud et al. 2014
Bowland Fells	NW England	11,470 Or 13,900 (1998) (Stroud et al. 2001)	1993		18,518 4,575	2001 2008-2012	SMP database Stroud et al. 2014
Morecambe Bay	NW England	22,000 (Stroud et al. 2001)	1996		12,100 11,988 10,354 10,670 9,829 8,130 4,987	2006 2007 2008 2009 2010 2011 2012	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Ribble and Alt Estuaries		1,800 (1993)	1995	The 2012 count used a new method and may not be a real increase from 2008	4,150 3,348 4,117 8,267	1998 2003 2008 2012	SMP database SMP database SMP database SMP database
Skomer and Skokholm	Wales	20,300 (1993-1997)	1982		12,660 12,780 12,690 10,890 9,640	2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database
Isles of Scilly	SW England	3,608 (1999)	2001		3,400 3,333	2006 2006	SMP database Stroud et al. 2014

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

12.10 BDMPS

UK waters can be split into two spatial BDMPS for lesser black-backed gulls, the UK North Sea and Channel, and the UK western waters (Figure 12.8). This split is based on the fact that while some lesser black-backed gulls from colonies in western Britain move into the North Sea during autumn migration, many tend to move southwards in autumn through UK western waters whereas birds from North Sea colonies tend primarily to move southwards through the North Sea. In addition, birds from overseas are likely to show a tendency to occur more in one side of the UK than the other, with birds from continental Europe more frequent in the North Sea than in western waters. There is a need to define three distinct seasonal BDMPS in each of these spatial units – autumn migration (August to October), winter (November to February), and spring migration (March and April). Numbers are much smaller in winter than during the migration periods.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 36 to 41.

Based on evidence reviewed in sections 12.5, 12.6 and 12.7, in autumn in the UK North Sea and Channel, the BDMPS is estimated to include 100% of adults and 70% of immatures from colonies on the UK North Sea coast, 50% of adults and 40% of immatures from UK colonies in western Scotland, Northern Ireland and NW England, 30% of birds from colonies in Wales and 10% of adults and 5% of immatures from the Isles of Scilly (Appendix A Table 36). The BDMPS is also estimated to include birds from several overseas populations; 20% of adults and 10% of immatures from Iceland, 30% of adults and 10% of immatures from Norway, 40% of adults and 20% of immatures from Faroe, 10% of adults and 5% of immatures from Sweden, Denmark and Ireland, 5% of adults and 2.5% of immatures from The Netherlands. These proportions result in an estimated BDMPS of 209,007 birds in the UK North Sea and Channel in autumn, 146,137 from the UK and 62,870 from overseas.

Based on evidence reviewed in sections 12.5, 12.6 and 12.7, in autumn in UK western waters, the BDMPS is estimated to include no adults from UK North Sea colonies but 10% of immatures from those sites, 50% of adults and 40% of immatures from colonies from west Scotland, Northern Ireland and NW England, 70% of adults and 40% of immatures from colonies in Wales, 90% of adults and 60% of immatures from colonies in SW England (Appendix A Table 37). The BDMPS is also estimated to include birds from several overseas populations; 20% of adults and 10% of immatures from Iceland, 10% of adults and 5% of immatures from Norway, 40% of adults and 20% of immatures from Faroe, 5% of adults and 2% of immatures from Sweden and Denmark, 40% of adults and 20% of immatures from Ireland, 2.5% of adults and 1% of immatures from The Netherlands. These proportions result in an estimated BDMPS of 163,304 birds in the UK North Sea in autumn, 120,205 from the UK and 43,099 from overseas.

Based on evidence reviewed in sections 12.5, 12.6 and 12.7, in winter in the UK North Sea and Channel, the BDMPS is estimated to include 50% of adults and 5% of immatures from colonies on the UK North Sea coast, 10% of adults and 1% of immatures from UK colonies in western Scotland, Northern Ireland, Wales and W England (Appendix A Table 38). The BDMPS is also estimated to include birds from several overseas populations; 5% of adults but no immatures from Iceland, Norway, and Faroe, 1% of adults but no immatures from Sweden, Denmark and Ireland, 0.5% of adults but no immatures from The Netherlands. These proportions result in an estimated BDMPS of 39,314 birds in the UK North Sea and Channel in winter, 31,590 from the UK and 7,724 from overseas.

Based on evidence reviewed in sections 12.5, 12.6 and 12.7, in winter in UK western waters, the BDMPS is estimated to include no birds from UK North Sea colonies, 20% of adults and 5% of immatures from colonies from west Scotland, Northern Ireland, Wales and W England (Appendix A Table 39). The BDMPS is also estimated to include birds from several overseas populations; 5% of adults but no immatures from Iceland, 2% of adults but no immatures from Norway, 5% of adults but no immatures from Faroe, 1% of adults but no immatures from Sweden and Denmark, 20% of adults and 5% of immatures from Ireland, 0.5% of adults but no immatures from The Netherlands. These proportions result in an estimated BDMPS of 41,159 birds in the UK North Sea in winter, 33,533 from the UK and 7,626 from overseas.

Based on evidence reviewed in sections 12.5, 12.6 and 12.7, in spring in the UK North Sea and Channel, the BDMPS is estimated to include 100% of adults and 70% of immatures from colonies on the UK North Sea coast, 50% of adults and 40% of immatures from UK colonies in western Scotland, Northern Ireland and NW England, 30% of birds from colonies in Wales and 10% of adults and 5% of immatures from the Isles of Scilly (Appendix A Table 40). The BDMPS is also estimated to include birds from several overseas populations; 10% of adults and 5% of immatures from Iceland, 30% of adults and 10% of immatures from

Norway, 20% of adults and 10% of immatures from Faroe, 10% of adults and 5% of immatures from Sweden, Denmark and Ireland, 5% of adults and 2.5% of immatures from The Netherlands. These proportions result in an estimated BDMPS of 197,483 birds in the UK North Sea and Channel in spring, 146,137 from the UK and 51,346 from overseas.

Based on evidence reviewed in sections 12.5, 12.6 and 12.7, in spring in UK western waters, the BDMPS is estimated to include no adults from UK North Sea colonies but 10% of immatures from those sites, 50% of adults and 40% of immatures from colonies from west Scotland, Northern Ireland and NW England, 70% of adults and 40% of immatures from colonies in Wales, 90% of adults and 60% of immatures from colonies in SW England (Appendix A Table 41). The BDMPS is also estimated to include birds from several overseas populations; 20% of adults and 10% of immatures from Iceland, 10% of adults and 5% of immatures from Norway, 40% of adults and 20% of immatures from Faroe, 5% of adults and 2% of immatures from Sweden and Denmark, 40% of adults and 20% of immatures from Ireland, 2.5% of adults and 1% of immatures from The Netherlands. These proportions result in an estimated BDMPS of 163,304 birds in the UK North Sea in spring, 120,205 from the UK and 43,099 from overseas.

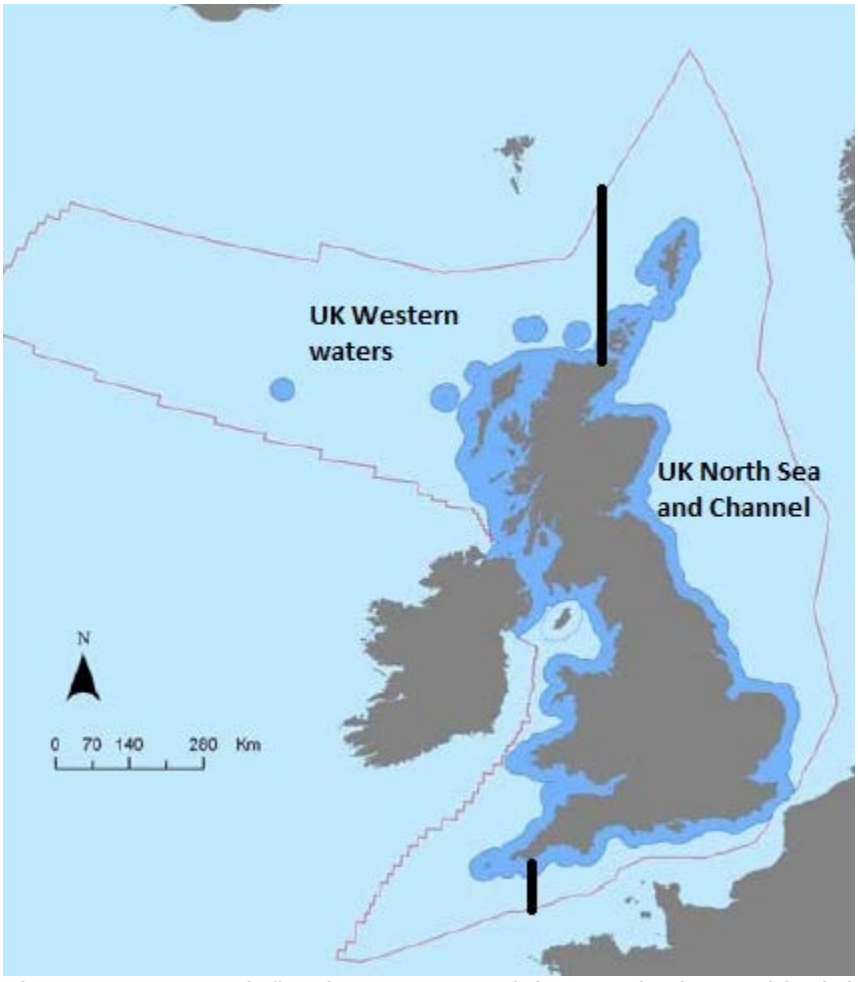


Figure 12.8. Two defined BDMPS spatial areas for lesser black-backed gull: 'UK North Sea and Channel' and 'UK Western waters'.

12.11 Proportions of UK SPA birds in each BDMPS

These proportions can be estimated directly from data in Appendix A Tables 36 to 41. For example, in the UK North Sea and Channel autumn migration BDMPS (Appendix A Table 36), there are 209,007 birds in the BDMPS, of which 29,572 are adults from UK SPA

populations, giving a percent of 14%. In contrast, in the UK western waters autumn migration BDMPS (Appendix A Table 37), there are 163,304 birds in the BDMPS, of which 38,228 are from UK SPA populations, giving a percent of 23%.

12.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Most SPA populations of lesser black-backed gulls are in southern Britain, and the northernmost SPA populations (Forth Islands in the east, Ailsa Craig and Rathlin Island in the west) hold only 1,608 pairs, 183 pairs and 107 pairs respectively (Table 12.1), so the proportions of UK SPA birds in the northern parts of the North Sea and the West of Scotland will be lower than in the southern parts. During the migration seasons and during winter, birds are likely to be well mixed with a large number of UK SPA, UK non-SPA, and overseas populations represented. As a result, proportions of birds within each BDMPS that are adults from UK SPA populations will be likely to be fairly consistent across much of each BDMPS spatial area, apart from a likely tendency for the proportion of UK SPA birds to be lower in the northern parts of each BDMPS range.

13. HERRING GULL *Larus argentatus*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (September to February)
Overseas	555,000	145,696
UK	543,000	494,114
Total	1,098,000	639,810

Non-breeding season BDMPS (September to February)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
UK North Sea and Channel	466,511	135,130	331,381
UK Western waters	173,299	10,566	162,733

Most UK herring gull SPA populations have been censused since Seabird2000. The JNCC seabird monitoring programme indicates a decline in breeding numbers since 2000, as do counts from several SPA colonies. Because a high proportion of breeding herring gulls in the UK are not in SPA colonies, up to date breeding numbers away from major SPA populations are less well known. Movements of breeding adults and of immatures in the UK have been studied in detail by individual colour ringing of birds in wintering areas and on migration, and have provided a fairly comprehensive picture of local movement patterns as well as connectivity with overseas populations. The key overseas population in the Barents Sea is thought to be approximately stable in numbers. Ringing studies abroad have also shown migrations of herring gulls from Faroe and Norway. Thus although there have not been geolocator tracking studies of herring gulls, the colour ringing work in the late 20th century does provide a good understanding of herring gull movements. BDMPS contributions from UK and overseas populations are coded amber.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 42 and 43.

13.1 Breeding range and taxa

The herring gull breeds across the Western Palearctic, with two subspecies. Birds breeding in Britain and Ireland are the endemic subspecies *argenteus*. Seabird 2000 reported 132,000 pairs in the UK and 5,500 pairs in Ireland (Mitchell et al. 2004). Elsewhere in northern Europe, birds are of the nominate subspecies *argentatus*. Herring gulls show clinal variation in size, with birds from northern Europe noticeably larger than those from the British Isles. They also show variation in the grey shade of the mantle and upperwing, and variation in wing tip pattern. These variations can be used to infer origins of individual birds at least in terms of broad geographical regions; in particular, adult birds from northern colonies can be

identified in the field when alongside British herring gulls, from differences in size and colour, though differences are not quite so obvious in juveniles and immatures.

13.2 Non-breeding component of the population

Herring gulls start to breed when 4 years old (BTO Birdfacts). Adult survival rate is 0.88 (BTO Birdfacts; Pons and Migot 1995), juvenile survival 0.63 up to age 4 years (BTO Birdfacts) and mean productivity is 0.936 chicks per pair (JNCC database, n=136 measurements). To obtain a stable population, survival of immatures was adjusted to 0.6 for juveniles, 0.7 for 1-year olds, 0.75 for 2-year olds, 0.83 for 3-year olds. The model population comprised 48% adults, 22% juveniles and 30% older immatures. There are 1.09 immatures per adult. The use of an alternative adult survival rate (for example derived from studies at Skomer http://jncc.defra.gov.uk/page-2886) would only alter this ratio very slightly.

13.3 Phenology

Although most adults remain close to their breeding sites throughout the year, few adults remain at colonies after August, with modal departure in August (Pennington et al. 2004; Forrester et al. 2007). However, as a partial migrant species in the UK, some adults remain close to their colony throughout the year. Autumn dispersal/migration starts in August (Wernham et al. 2002; Forrester et al. 2007) or mid-August (Cramp et al. 1977-94). Peak autumn migration occurs in July-December (Brown and Grice 2005), September-October (Forrester et al. 2007; Pennington et al. 2004), or October (Cramp et al. 1977-94; Wernham et al. 2002). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late October to late December (Figure 13.1). Trektellen sites (predominantly in east and south-east England) may observe mostly herring gulls arriving from north Norway rather than dispersing birds from UK colonies, but timing of autumn movements appears not to differ much between UK and north Norwegian populations (Stanley et al. 1981; Horton et al. 1983; Brown and Grice 2005). Autumn migration is completed by November (Forrester et al. 2007), early December (Cramp et al. 1977-94), or December (Wernham et al. 2002).

Spring migration starts in January (Wernham et al. 2002; Forrester et al. 2007) or mid-February (Cramp et al. 1977-94). Peak spring migration occurs in January (Pennington et al. 2004), January-April (Forrester et al. 2007), or March-April (Cramp et al. 1977-94; Wernham et al. 2002; Brown and Grice 2005). Peak rate of change in numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in February-April (Figure 13.1). Spring migration is completed by early May (Cramp et al. 1977-94) or May (Wernham et al. 2002; Forrester et al. 2007).

The first spring records of herring gull in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 1 January and the last records were predominantly at 31 December, as large numbers of herring gulls overwinter, but peak autumn migration was reported in October in most years, and peak spring migration was reported in January-March if detected at all which it was not in most years. Birds re-occupy colonies from early January, with modal return in early March (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007).

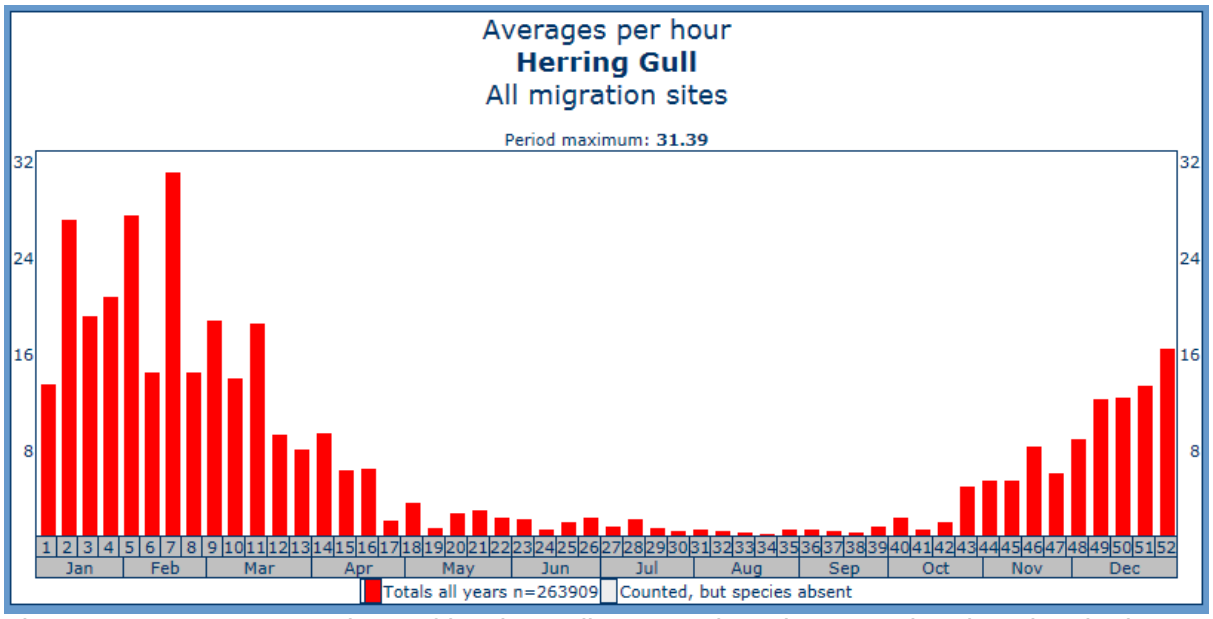


Figure 13.1. Average numbers of herring gulls counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as April-August, non-breeding season September-March. However, from the data reviewed above, a more appropriate definition would be breeding season March-August, non-breeding season September-February.

13.4 Defined seasons:

- UK Breeding season — March-August
- Post-breeding migration in UK waters — August-November
- non-breeding season — September-February (**non-breeding BDMPS**)
- Return migration through UK waters — January-April
- Migration-free breeding season — May-July
- Migration-free winter season — December

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for herring gull:

Non-breeding season BDMPS (September-February).

13.5 Movements of birds from the UK population

Herring gulls in Britain and Ireland do not migrate, and show only limited dispersal. Most adults remain close to their breeding sites throughout the year. Young birds move further than adults, but the median distance between ringing site and recovery site for all UK ringed herring gulls (so predominantly ringed as chicks in colonies) was only around 15 km (Wernham et al. 2002). Camphuysen (2013) found that successful breeders abandoned the colonies in The Netherlands in July-August, when their young were about 50 days old, and that southward autumn movement started first in immatures, then in adults, and last in juveniles, suggesting that post-fledging care of juveniles was mostly minimal. At UK colonies, dispersal after breeding can be evident from August onwards and while birds can move in all directions the autumn movements tend to be predominantly southwards, but lead to little increase in distance between ringing and recovery site until October. A measureable

but small average distance is evident in ring recoveries until March, but by April virtually all recoveries of adults are at or very close to the colony, although immature birds may be somewhat more widely distributed (Wernham et al. 2002). Studies on refuse tips in north-east England found that colour ringed herring gulls originated from the whole east coast of Scotland as well as local birds from NE England. Adults started to arrive from late July (presumably these were failed breeders) with peak passage in September-October and some individuals not arriving until December, with a tendency for individuals to show the same seasonal pattern in successive years (Wernham et al. 2002). Herring gulls generally tend to remain close to coasts, occurring at rather low density in pelagic waters. Although herring gulls may move along coasts or sometimes across water, movements between east and west coasts of the UK are surprisingly limited (Wernham et al. 2002). Populations to the west of the UK are therefore unlikely to mix much with populations to the east of the UK.

13.6 Movements of birds from overseas into UK waters

Many nominate *argentatus* birds, especially those from furthest north, can be identified in the field from their considerably larger body size and plumage features (darker mantle, white tip to outermost primary). In winter, those birds tend to be seen in largest numbers in eastern Britain (Coulson et al. 1984). Birds from the nominate subspecies mostly occur in the UK from September to February (Wernham et al. 2002). Ringing suggests that very few of those birds come from Iceland (Wernham et al. 2002). Ringing in Faroe has resulted in two recoveries of birds ringed as chicks and subsequently recovered in their first winter in the UK (Hammer et al. 2013), indicating that at least some young birds from Faroe winter in the UK. However, these come from a relatively small population (1,500 pairs; Hammer et al. 2013). Much larger numbers arrive from the Barents Sea coast of north Norway and north Russia (Wernham et al. 2002), where there are around 126,000 pairs (Anker-Nilssen et al. 2000). Those birds obviously carry out a long-distance migration with extensive travel across the sea from Norway to Scotland, but perhaps surprisingly they very rarely occur in west Britain, tending to remain on the east coast of the UK from Shetland to SE England (Wernham et al. 2002). The Barents Sea population of herring gulls is considered to be partially migratory, with some adults remaining in the Barents Sea throughout the year, but some adults and a higher proportion of immatures migrate during October to winter in the North Sea. Birds from northern Norway winter further south than birds from southern Norway (Haftorn 1971), so Norwegian birds in UK waters are almost all of northern Norwegian origin. Large numbers of herring gulls (many thousands) overwinter along the coast of southern Norway (Petersen et al. 2011), but those birds are probably mostly local breeders that remain in the same area throughout the year, possibly with some birds from north Norway too. While birds from the Norwegian sector of the Barents Sea predominantly migrate along the Norwegian coast, birds from the Russian sector of the Barents Sea (including the White Sea) mostly migrate through the Baltic Sea. Some of these reach the North Sea, while others winter further east (Anker-Nilssen et al. 2000). Herring gulls breeding in The Netherlands are largely sedentary, with many adults remaining within a few km of their colony through winter (Camphuysen 2013). The limited dispersal of herring gulls from colonies in The Netherlands apparently does not normally involve movements to the UK since only 3 sightings out of over 86,000 movements of colour ringed herring gulls from colonies in The Netherlands were made in the UK (Camphuysen et al. 2011).

13.7 Numbers in UK waters

Musgrove et al. (2013) report that there are 730,000 in Britain in winter, 740,000 in UK in winter, but it appears that these totals do not include birds at sea except where they were visible from land. From surveys in 2007 and 2008, Fauchald and Tveraa (2009) reported mean densities at sea of 9.7 to 13.6 birds per km^2 in the Norwegian Sea in spring/summer, and 1.8 to 6.4 birds per km^2 in the Barents Sea in autumn. Nearly 1,000,000 herring gulls are in the entire North Sea in winter (November to February) dispersed throughout the North Sea but many of these birds are not in UK waters (Skov et al. 1995) (although these data are

now rather out of date). About 175,000 more winter in the Wadden Sea (Camphuysen 2013). Lack (1986) estimated that about 500,000 herring gulls winter inland or on coasts in Britain and Ireland, with about 122,000 of these in England (Brown and Grice 2005). There were estimated to be 63,780 birds at inland roosts in England in January 1993 and 192,846 at coastal roosts (Burton et al. 2003; Brown and Grice 2005; Burton et al. 2013). Forrester et al. (2007) suggest that there are well over 91,000 herring gulls from the UK population in Scotland, in mid-winter, in terrestrial habitats, but numbers that may be at sea at that time in addition to this total were not estimated, and that count did not include herring gulls in Shetland, Orkney, Western Isles or several parts of northern Scotland, so this number is clearly a large underestimate. In addition, Forrester et al. (2007) estimated that between 5,000 and 20,000 Scandinavian herring gulls are in Scotland in winter, but again this estimate seems to be based mainly on data from terrestrial sites rather than from marine habitats, and is likely to be an underestimate of the total.

In March-April most central areas of the North Sea are vacated by herring gulls, with concentrations found in the Southern Bight and German Bight, the Skagerrak/Kattegat, and in Shetland to NE Scotland (Camphuysen 2013). In summer and early autumn, herring gull numbers in the North Sea are low, showing a coastal distribution related to breeding colony locations (Camphuysen 2013). In winter, herring gulls show a strong association with the distribution of fishing vessels, congregating in areas where fisheries discards are available (Camphuysen et al. 1995). Thus the numbers and distribution of herring gulls in UK waters in winter are likely to vary in response to changes in fisheries activity.

13.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the NW Europe population, comprising 940,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 705,000-799,000 pairs. Kober et al. (2010) did not present an estimated biogeographic population for this species. Populations with connectivity to UK waters sum to 262,500 pairs, with the UK population almost exactly half of this. Thus the biogeographic population including immatures as well as adults may number about 1,098,000 birds, with 543,000 from UK and 555,000 from overseas. However, only part of the large Barents Sea population comes into UK waters in winter, so UK birds will tend to outnumber birds from overseas populations during migration periods and midwinter. The total numbers in UK waters in the non-breeding season (September to February) sum to a total of about 640,000 birds, 494,000 from UK and 146,000 from overseas.



Figure 13.2. Breeding population origins of herring gulls in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

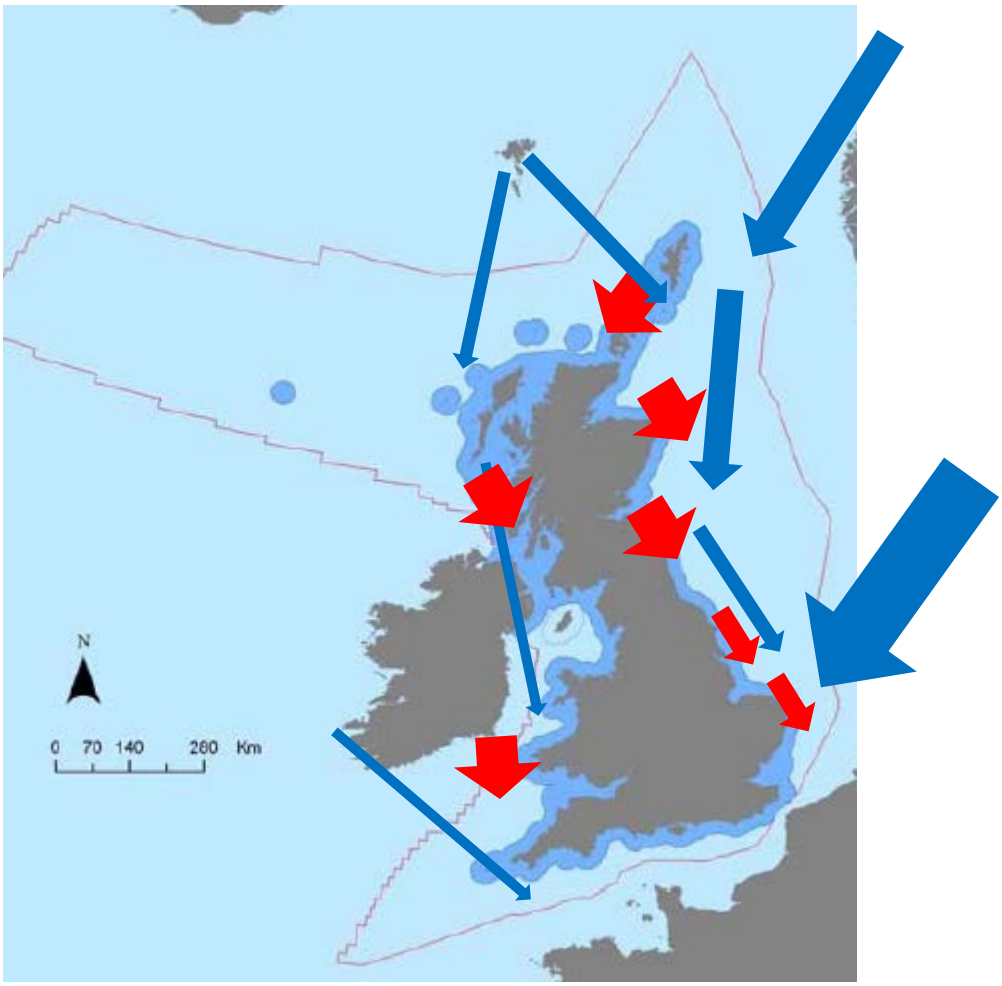


Figure 13.3. Main movements of herring gulls from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

13.9 Proportion of BDMPS from UK breeding SPAs

The 12 SPAs with breeding herring gulls as a feature together held 54,650 pairs at designation, estimated to represent ca. 32% of the British breeding population (Stroud et al. 2001) (although this misses some of the inland breeding colonies so probably rather overestimates the proportion breeding on SPAs; G Mudge in litt.). Herring gull numbers have declined considerably since these SPAs were designated, and as with other declining seabird populations, the decreases have been especially large in the largest populations, which are the SPAs. Stroud et al. (2014) estimated that the UK SPA suite for breeding herring gulls held 12.5% of the GB population in 1999-2011, and since numbers have declined further at some of the SPAs where they used data from 1999-2003, this percentage has almost certainly decreased further and may now be around 11% based on more up to date data in Table 13.1.



Figure 13.4. The SPA suite for herring gull. These SPA populations are listed in Table 13.1.

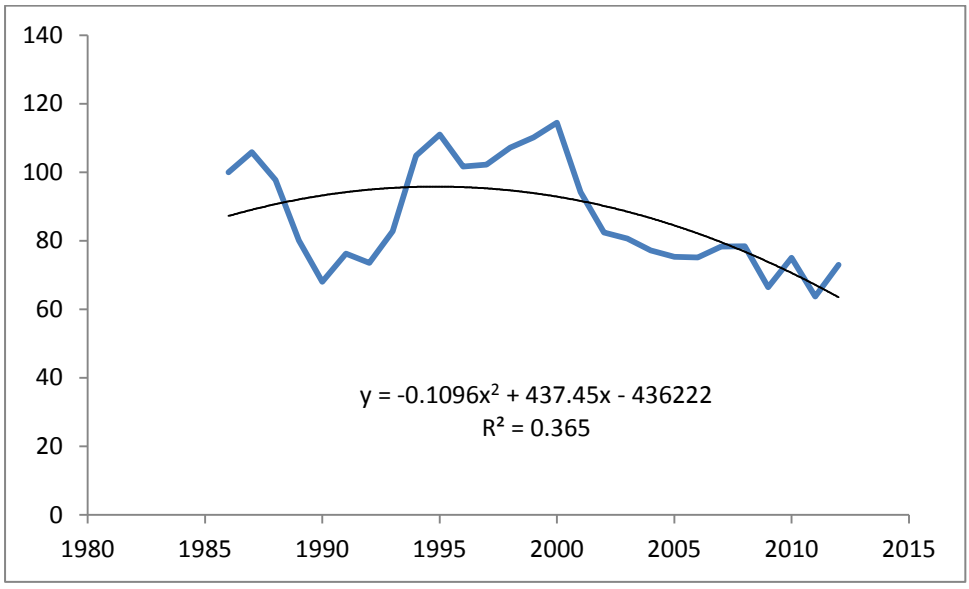


Figure 13.5. Trend in the herring gull breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

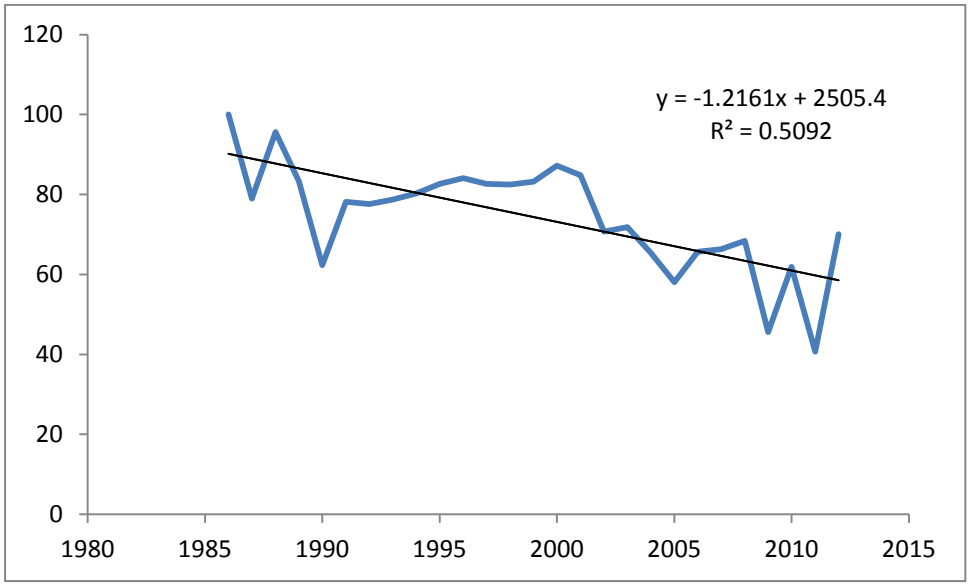


Figure 13.6. Trend in the herring gull breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

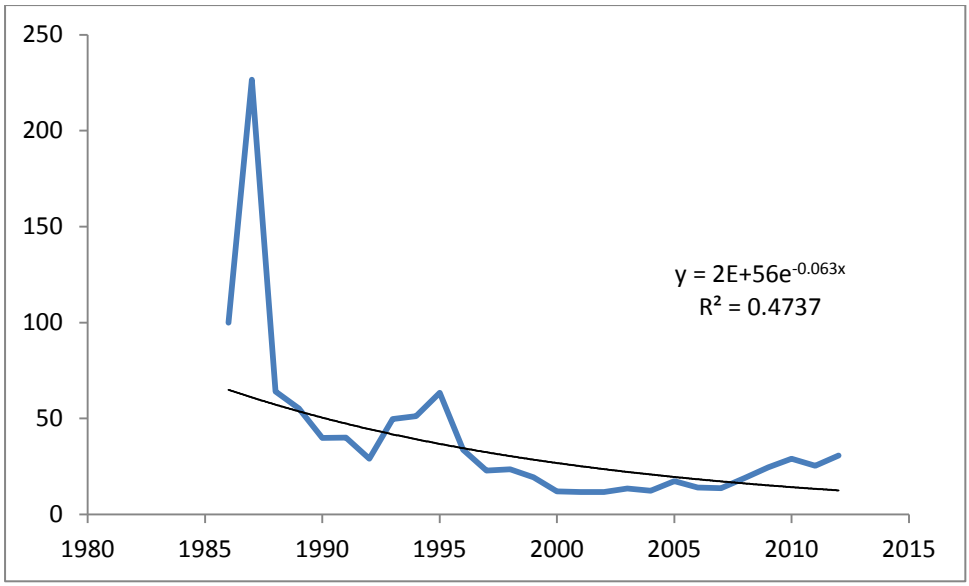


Figure 13.7. Trend in the herring gull breeding population index in Northern Ireland from 1986-2012. Data from JNCC seabird population monitoring database.

Table 13.1. The UK SPA suite for breeding herring gulls.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
East Caithness Cliffs	N Scotland	9,370 (1986)	1996	Declined 1999	3,393	1999	Seabird2000
Troup, Pennan and Lion's Heads	NE Scotland	4,200 (1995)	1997	No change 2007	1,951 1,687 1,597	2001 2007 2007	Lewis et al. 2012 Lewis et al. 2012 SCM database
Buchan Ness to Collieston Coast	NE Scotland	4,292	1998	No change 2007	3,079 3,114	2007 2010	SCM database Lewis et al. 2012

Fowlsheugh	NE Scotland	3,190	1992	Declined 1999	122 214 259	2008 2009 2012	Lewis et al. 2012 Lewis et al. 2012 SCM database
Forth Islands	E Scotland	6,600 (1985)	1990	Maintained 2001	5,026 5,100 2,827	2002 2004 -12 2005 -09	Lewis et al. 2012 SCM database Stroud et al. 2014
St Abb's Head to Fast Castle	SE Scotland	1,160	1997	Declined 2002	541 647 220 266 239	2000 2000 2011 2012 2013	Seabird2000 Stroud et al. 2014 SCM database SCM database SCM database
Flamborough Head & Bempton Cliffs	E England	1,110 (1987)	1993		721 533 495	2000 2008 2010	SCM database SCM database SCM database
Alde-Ore Estuary	SE England	6,050 (Stroud et al. 2001)	1996	These counts are for Orfordness only and exclude Havergate	6,750 2,575 2,000 1,000 1,000 800	2000 2002 2003 2004 2005 2006	SCM database SCM database SCM database SCM database SCM database SCM database
UK Western waters							
Canna and Sanday	Inner Hebrides	1,391	1998	Declined 2001	70 63	2010 2011	Lewis et al. 2012 Lewis et al. 2012
Ailsa Craig	W Scotland	2,250 (1987)	1990	Declined 2010	131 82 129	2010 2012 2013	Lewis et al. 2012 SCM database SCM database
Rathlin Island	N Ireland	4,037	1999		14 5 28 23	1999 2007 2011 2011	SCM database SCM database SCM database Stroud et al. 2014
Morecambe Bay	NW England	11,000	1996		3,225 3,040 2,246 2,094 1,734	2008 2009 2010 2011 2012	SCM database SCM database SCM database SCM database SCM database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

13.10 BDMPS

UK waters can be split into two spatial BDMPS for herring gulls; UK North Sea and Channel waters, and UK western waters. Although some birds move between these two areas, there is a distinct tendency for birds to remain in one or other of these two areas with little interchange. Also, birds from the Barents Sea tend to migrate into the North Sea in large numbers, but very few of those birds enter UK western waters. Population sizes in these two spatial BDMPS are essentially the same for the migration periods (once birds from overseas have reached UK waters and until they depart in spring) and winter, so there is no requirement to split these into separate temporal units.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 42 and 43.

Based on evidence reviewed in sections 13.5, 13.6 and 13.7, in the non-breeding season (September to February) in the UK North Sea and Channel, the BDMPS is estimated to include 99% of adults and 95% of immatures from colonies on the UK North Sea coast, 5% of adults and 10% of immatures from UK colonies in western Scotland, Northern Ireland, Wales and W England (Appendix A Table 42). The BDMPS is also estimated to include birds from three overseas populations; 20% of adults and 30% of immatures from the Barents Sea, 20% of adults and 30% of immatures from Faroe, 2% of adults and 5% of immatures from Ireland. These proportions result in an estimated BDMPS of 466,511 birds in the UK North Sea and Channel in the non-breeding season, 331,381 from the UK and 135,130 from overseas.

Based on evidence reviewed in sections 13.5, 13.6 and 13.7, in the non-breeding season (September to February) in the UK western waters, the BDMPS is estimated to include 0.1% of adults and 0.1% of immatures from colonies on the UK North Sea coast, 80% of adults and 70% of immatures from UK colonies in western Scotland, Northern Ireland, Wales and W England (Appendix A Table 43). The BDMPS is also estimated to include birds from three overseas populations; 0.1% of adults and 0.5% of immatures from the Barents Sea, 20% of adults and 30% of immatures from Faroe, 30% of adults and 40% of immatures from Ireland. These proportions result in an estimated BDMPS of 173,299 birds in UK western waters in the non-breeding season, 162,733 from the UK and 10,566 from overseas.

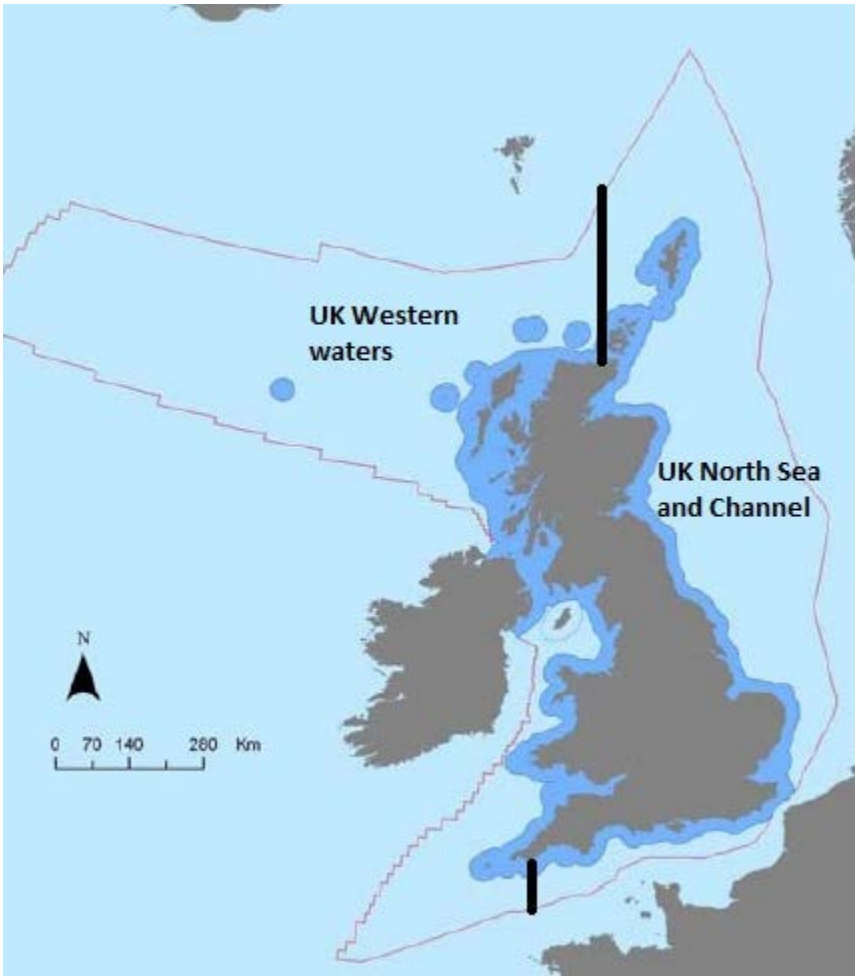


Figure 13.8. Two defined BDMPS spatial areas for herring gull: 'UK North Sea and Channel' and 'UK Western waters'.

13.11 Proportions of UK SPA birds in each BDMPS

About 11% of the UK adult herring gull population breeds in the UK SPA suite for breeding herring gull. Given that the SPAs for herring gull are distributed in a way that reflects fairly closely the breeding distribution of the species in the UK (Figure 13.4), this will probably apply in all areas. However the proportion will be diluted by the presence of immature birds and by the presence of birds from overseas populations. The proportion of birds in each BDMPS that are adults from UK SPA populations can be estimated directly from Appendix A Tables 42 and 43. For example, the UK North Sea and Channel non-breeding season BDMPS comprises 466,511 birds in total, of which 25,389 are adults from UK SPA populations, giving an estimate of 5.4% being adults from UK SPAs.

13.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Because adult herring gulls from UK colonies tend to remain close to their colony throughout the year, there is likely to be a tendency for SPA birds to be aggregated near the SPA sites, although immature birds will disperse more widely and be more mixed. There is some evidence to suggest that herring gulls from the Barents Sea population tend to be more marine than UK herring gulls during migration periods and winter, so that birds at sea may include a higher proportion of 'foreign' herring gulls while birds in terrestrial sites may include a higher proportion of UK herring gulls, and so also a higher proportion of birds from UK SPA populations than found at sea. However, this difference in local distribution of birds has not been quantified so cannot be assessed in any detail.

14. GREAT BLACK-BACKED GULL *Larus marinus*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (September to March)
Overseas	163,000	76,492
UK	72,000	67,029
Total	235,000	143,521

Non-breeding season BDMPS (September to March)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
UK North Sea	91,399	62,736	28,663
UK West of Scotland	34,380	9,677	24,703
UK South-west & Channel	17,742	4,079	13,663

Slightly more than half of the UK great black-backed gull SPA populations have been censused since Seabird2000, so breeding numbers in these large colonies are known in some cases but rather uncertain in others. The JNCC seabird monitoring programme indicates a decline in breeding numbers since 2000, as do counts from several SPA colonies. Because a high proportion of breeding great black-backed gulls in the UK are not in SPA colonies, up to date breeding numbers away from major SPA populations are less well known. Movements of breeding adults and of immatures in the UK have been studied in detail by individual colour ringing of birds in wintering areas and on migration, and have provided a fairly comprehensive picture of local movement patterns as well as connectivity with overseas populations. The key overseas population in the Barents Sea is thought to be approximately stable in numbers. Ringing studies abroad have also shown migrations of great black-backed gulls from Faroe and Norway. Although there have not been geolocator tracking studies of great black-backed gulls, the colour ringing work in the late 20th century does provide a good understanding of great black-backed gull movements, and these appear to be consistent from year to year. BDMPS contributions from UK and overseas populations are therefore coded amber overall.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 44 to 46.

14.1 Breeding range and taxa

This Holarctic breeding species is monotypic, and there is no evidence to suggest that biometrics are useful in assessing origins of individuals.

14.2 Non-breeding component of the population

Great black-backed gulls start to breed when 4 years old (BTO Birdfacts). Adult survival rate is unknown (BTO Birdfacts), juvenile survival unknown (BTO Birdfacts) and mean productivity is 1.139 chicks per pair (JNCC database, n=132 measurements). To obtain a stable population, adult survival was set at 0.88 (the same as herring gull), survival of immatures was adjusted to 0.56 for juveniles, 0.67 for 1-year olds, 0.74 for 2-year olds, and 0.78 for 3-year olds. The model population comprised 44% adults, 25% juveniles and 31% older immatures. There are 1.26 immatures per adult.

14.3 Phenology

Breeding colonies in the UK are deserted by early September, with modal departure in late July or early August (Pennington et al. 2004; Forrester et al. 2007). Autumn dispersal/migration starts in July (Wernham et al. 2002), August (Forrester et al. 2007) or mid-August (Cramp et al. 1977-94). Peak autumn migration occurs in July-October (Brown and Grice 2005), September (Wernham et al. 2002), October (Pennington et al. 2004), September-October (Forrester et al. 2007), or September-November in Belgium (Vanermen et al. 2013) or throughout Europe (Cramp et al. 1977-94). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in November-December (Figure 14.1), suggesting that those sites recorded later arriving birds from north Norway rather than birds dispersing from UK colonies. Autumn migration is completed by November (Forrester et al. 2007), early December (Cramp et al. 1977-94) or December (Wernham et al. 2002).

Spring migration starts in January (Forrester et al. 2007), February (Wernham et al. 2002) or mid-February (Cramp et al. 1977-94). Peak spring migration occurs in January-February in Belgium (Vanermen et al. 2013), January in Shetland (Pennington et al. 2004) January-April (Forrester et al. 2007), late February in England (Brown and Grice 2005), February-March (Wernham et al. 2002), or March (Cramp et al. 1977-94). Peak rate of change in numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in January-March (Figure 14.1). Spring migration is completed by April (Wernham et al. 2002) early May (Cramp et al. 1977-94), or May (Forrester et al. 2007).

The first spring records of great black-backed gull in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 1 January and the last records were at 31 December, as large numbers of great black-backed gulls overwinter, while peak autumn migration was reported in October or November in most years, and peak spring migration was reported as not evident in most years. Birds re-occupy colonies from early February, with modal return in March (Forrester et al. 2007).

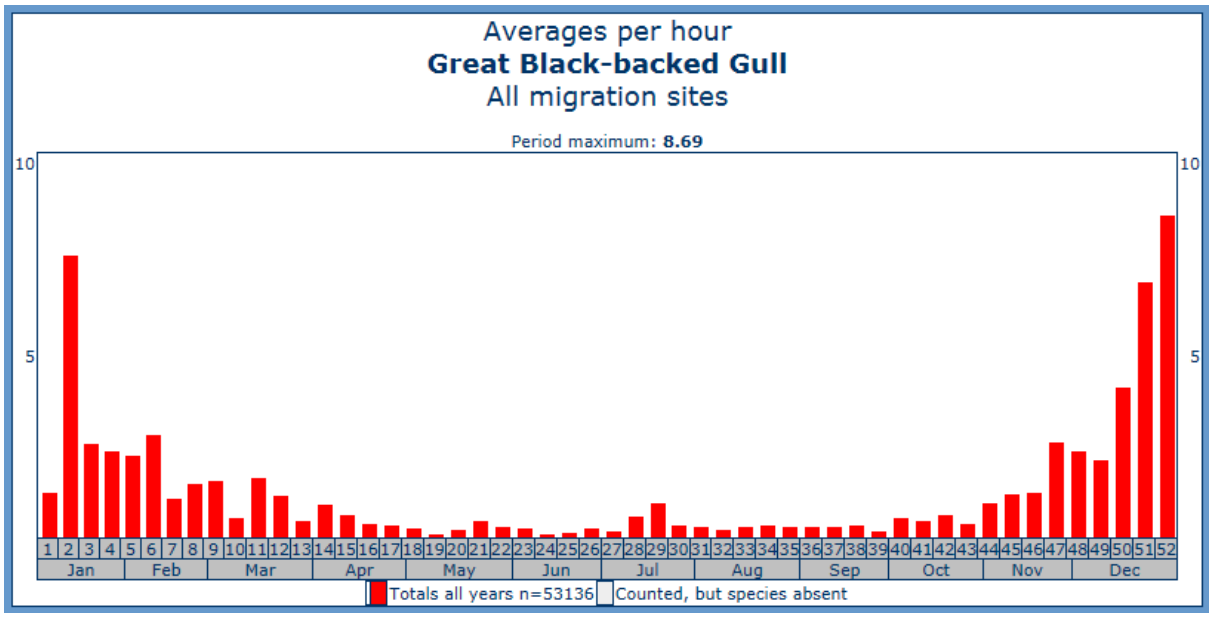


Figure 14.1. Average numbers of great black-backed gulls counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as April-August, non-breeding season September-March. From the data reviewed above, this appears to be an appropriate definition.

14.4 Defined seasons:

- UK Breeding season — late March-August
- Post-breeding migration in UK waters — August-November
- non-breeding season — September-March (**non-breeding BDMPS**)
- Return migration through UK waters — January-April
- Migration-free breeding season — May-July
- Migration-free winter season — December

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for great black-backed gull:

Non-breeding season BDMPS (September-March).

14.5 Movements of birds from the UK population

Adult great black-backed gulls in the UK are partial migrants, with adults being mainly sedentary or travelling only short distances from their breeding area. Some adults disperse short distances from colonies to winter mainly south or east of their colony, tending to return to the same wintering site each year (Coulson et al. 1984). Juveniles and older immatures disperse slightly further than adults; the median distance between colony and wintering area was 54 km for adults but 115 km for immatures ringed in Britain and Ireland (Wernham et al. 2002). Adults return to breeding areas in late winter. Birds ringed at colonies in the northern isles and north Scotland were mainly recovered close to the breeding areas where they were ringed, or down the east coast, a very few birds reaching the south coast of England or coast of the Netherlands or Belgium (Wernham et al. 2002). Very few of these birds crossed to the West coast of Britain or to Ireland. Birds ringed at colonies in the west of Scotland, northwest

of England or northern half of Ireland were mainly recovered close to the breeding areas where they were ringed, with a very few reaching the south coast of Ireland, Wales and SW of England. Extremely few birds from the west coast crossed Britain to reach the North Sea. Birds ringed at colonies in SW England, Wales, and the southern part of Ireland were mainly recovered close to the breeding areas where they were ringed, with a very few reaching France.

14.6 Movements of birds from overseas into UK waters

In contrast to the mainly sedentary nature of adult great black-backed gulls in Britain and Ireland, some birds from northern populations migrate long distances, especially to overwinter in the North Sea. Although large numbers breed in Iceland (15,000 to 20,000 pairs; Mitchell et al. 2004), and moderate numbers in Faroe (1,200 pairs; Hammer et al. 2013), these birds are predominantly sedentary (Wernham et al. 2002). Hammer et al. (2013) reported one recovery in the UK and three in Ireland of great black-backed gulls ringed in Faroe, all of which were recovered when less than a year old. Similarly, great black-backed gulls in southern Norway are considered to be mainly sedentary, most remaining in Norwegian waters throughout the year (Anker-Nilssen et al. 2000; Wernham et al. 2002). Foreign-ringed great black-backed gulls recovered in Britain and Ireland mainly originate from the north coasts of Norway and Russia. These birds begin arriving in July, mainly on the east coast of England (Wernham et al. 2002). Numbers peak in September (Wernham et al. 2002), then remain high through early winter until the return migration in February (Wernham et al. 2002). The Barents Sea population of great black-backed gulls, most of which breed along the north coast of Norway, is estimated at around 33,000 pairs (Anker-Nilssen et al. 2000). The Barents Sea great black-backed gull is a partial migrant. Some birds remain close to colonies all year round, while others migrate to winter in the North Sea. It is not clear what proportion of this population winters in the North Sea rather than in the Barents Sea or Norwegian Sea, or in the Caspian or Black Sea, but it is thought that the North Sea is their main wintering area (Anker-Nilssen et al. 2000). Most migrate along the Norwegian coast. Some migrate through the White Sea then along rivers to the Volga delta to winter in the Caspian or Black Sea. Some migrate overland between the White and Baltic Seas, then may continue to the North Sea (Anker-Nilssen et al. 2000). Southward movement is more extensive among immatures than among adults. Birds leave the breeding colonies in north Norway in August, but migration south mainly occurs in September-October (Anker-Nilssen et al. 2000). Adults arrive back at colonies in the Barents Sea in March-April (Anker-Nilssen et al. 2000). With a population in the UK of around 16,800 pairs, with many of these in colonies on the west of the British Isles rather than in the North Sea, the resident great black-backed gulls in the North Sea are likely to be outnumbered in winter by great black-backed gulls from northern Norway. There may be very small numbers of great black-backed gulls from southern Norway, Denmark, SW Sweden and France that visit UK waters, but these numbers appear to be so small relative to the large numbers from the Barents Sea and from the UK that they can be ignored as trivial.

14.7 Numbers in UK waters

Musgrove et al. (2013) report that there are 76,000 in Britain in winter, 77,000 in UK in winter, but these estimates only include birds at sea that could be counted from land, as well as birds onshore and at coastal roosts. From surveys in 2007 and 2008, Fauchald and Tveraa (2009) reported mean densities at sea of 4.8-11.3 birds per km^2 in the Norwegian Sea in spring/summer, and 0.5-1.4 birds per km^2 in the Barents Sea in autumn. Most migrants in English waters occur off east England, whereas most breeders in England are in Cornwall and the Scillies (Brown and Grice 2005). Some northern immatures remain in the southern North Sea all year round (Brown and Grice 2005). There were estimated to be 21,077 birds at inland roosts in England in January 1993 and 17,838 at coastal roosts (Burton et al. 2003; Brown and Grice 2005). Forrester et al. (2007) suggest that there are around 2,000 to 10,000 birds in Scotland during the migration seasons, and 7,500 to 10,000

in winter. However, these estimates appear to be based on counts of birds onshore rather than at sea. Skov et al. (1995) estimated that there are around 300,000 great black-backed gulls in the North Sea in winter (although these data are now rather out of date), with peak numbers in November to February (Stone et al. 1995). However, this number may be an overestimate because great black-backed gulls are attracted to boats (Kober et al. 2010). Since almost all UK great black-backed gulls winter in UK waters, there will be the 16,000 pairs from UK colonies (32,000 adults) plus associated immatures (about 40,000 of those) so about 72,000 birds. However, it is likely that about half of these are in waters west of the UK and half in the North Sea, as very few great black-backed gulls breed along the east coasts of England and Scotland except in the far north (Shetland, Orkney and Caithness). In contrast, the species breeds along most of the west coast of Scotland and in smaller numbers in Wales and west England. However, most of the SPA populations (the largest colonies) are in Orkney and north Scotland. In addition to birds from the UK, birds from Barents Sea colonies arrive in autumn, especially into the North Sea. It is uncertain how many of these winter in UK waters as some may winter in the Norwegian Sea (Anker-Nilssen et al. 2000), but there is evidence from colour ringing studies that relatively few from the Barents Sea winter in the west of Scotland. Count data suggest that the majority of birds in the North Sea in winter are likely to be from the Barents Sea. The Barents Sea population is 33,000 pairs and is apparently approximately stable (Anker-Nilssen et al. 2000, R.T. Barrett pers. comm.), so 66,000 adults plus about 83,000 immatures, so 149,000 birds. Given the estimate that up to 300,000 birds winter in the North Sea, it would seem likely that most birds from the Barents Sea population are in the North Sea in winter, as it would otherwise be impossible to reach such a large total.

14.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the European population, comprising 95,546 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 100,000-110,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 440,000 individuals. The biogeographic population with connectivity to UK waters comprises birds from the UK, Ireland, Faroe and Barents Sea (Figure 14.2). This sums to 235,000 birds (adults plus immatures), of which 72,000 are from UK and 163,000 from overseas populations. Substantial proportions of these populations occur in UK waters in the non-breeding period (September to March); the totals for UK waters are estimated at 143,000 birds, with 67,000 frrom UK and 76,000 from overseas populations.



Figure 14.2. Breeding population origins of great black-backed gulls in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

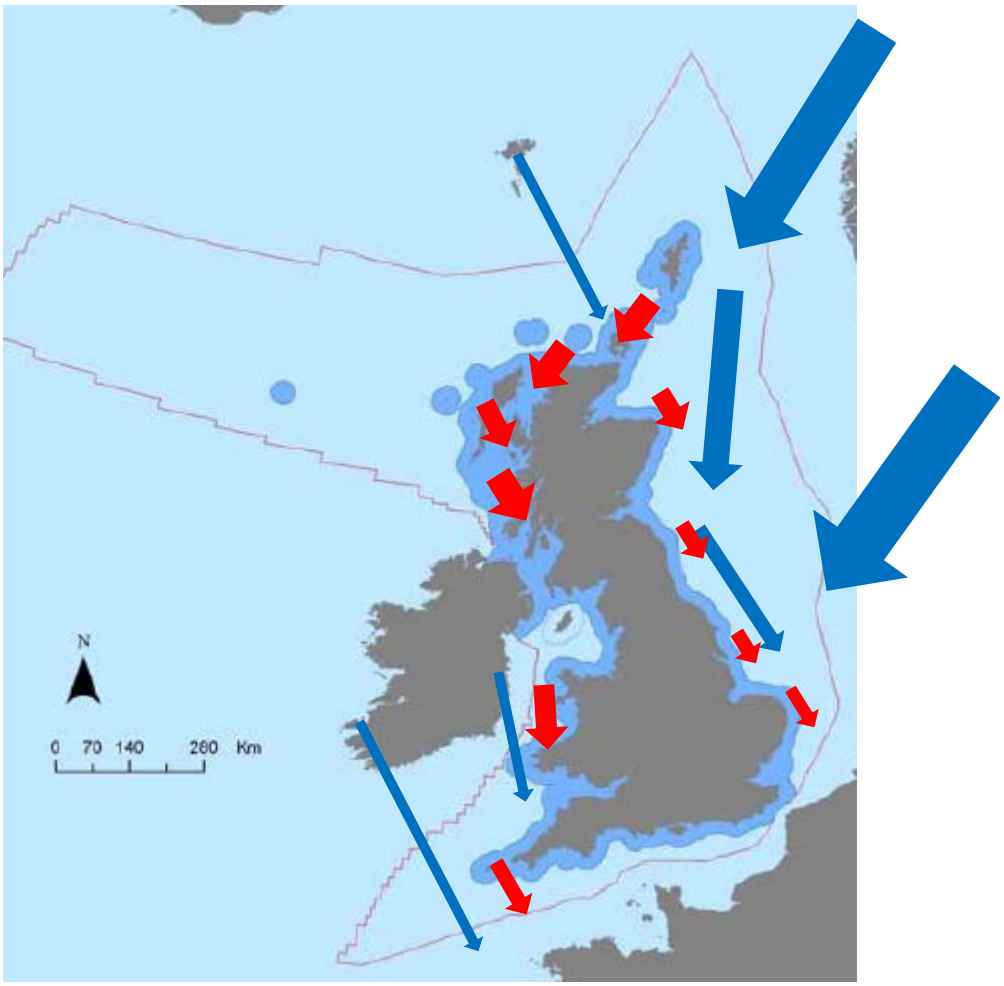


Figure 14.3. Main movements of great black-backed gulls from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

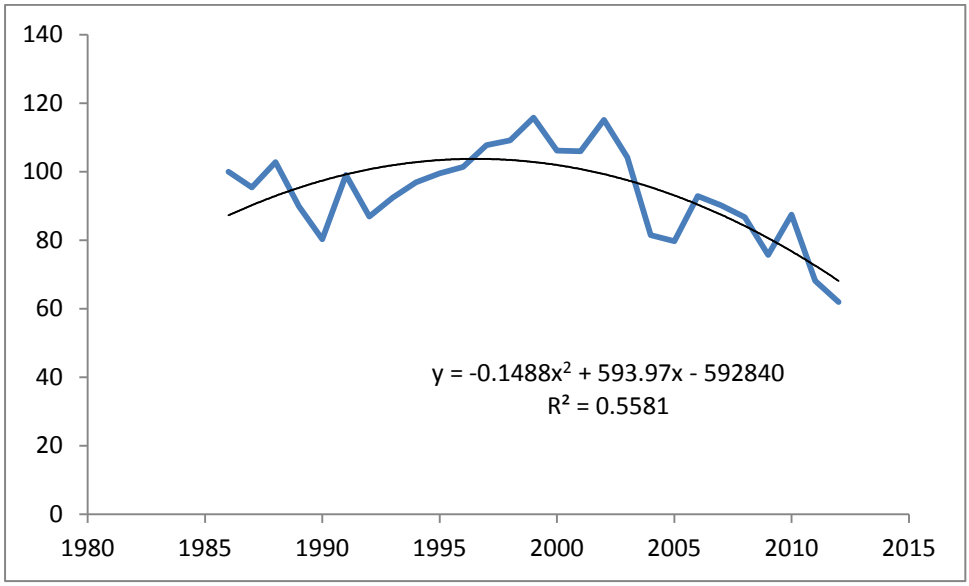


Figure 14.4. Trend in the great black-backed gull breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

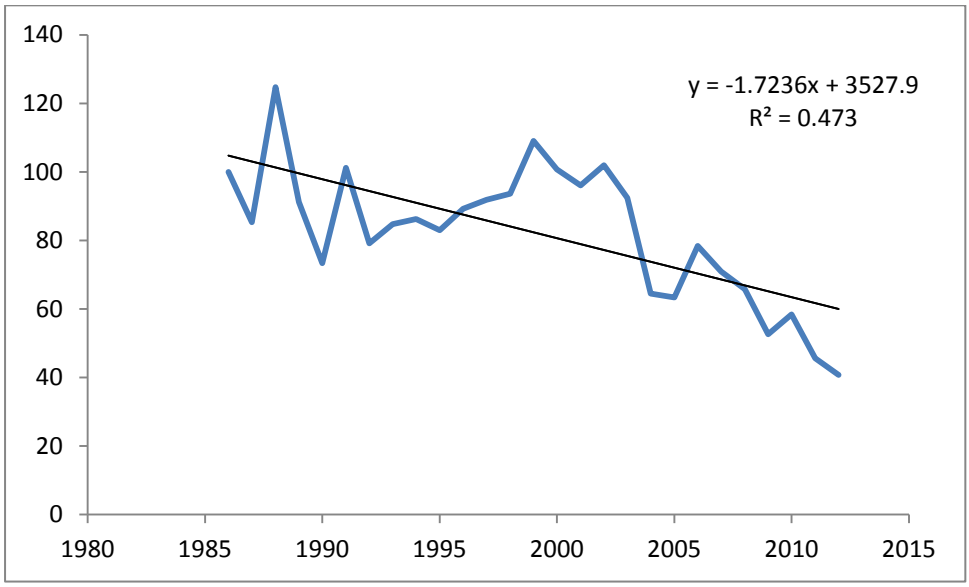


Figure 14.5. Trend in the great black-backed gull breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

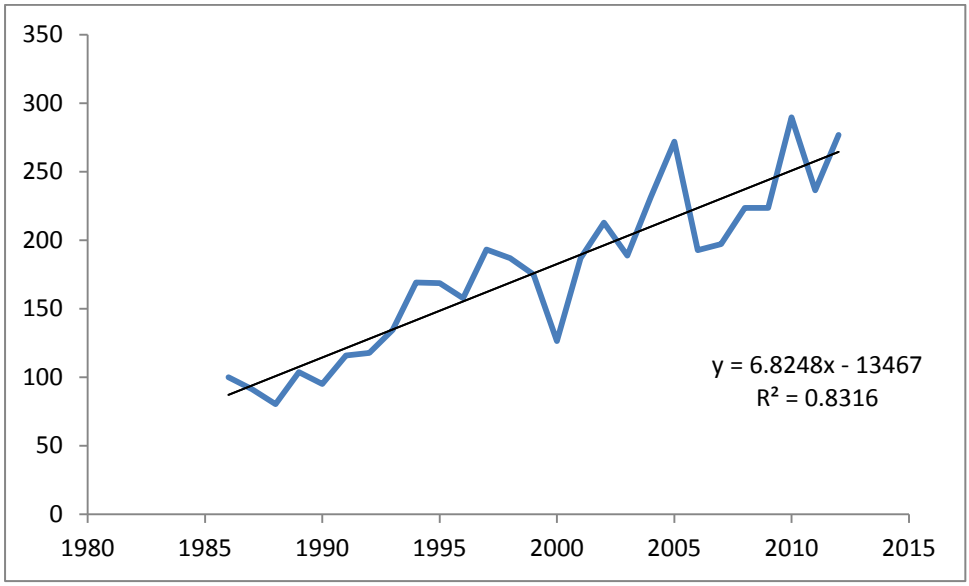


Figure 14.6. Trend in the great black-backed gull breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

14.9 Proportion of UK population from UK breeding SPAs

The six SPAs with breeding great black-backed gulls as a feature together held 4,457 pairs at designation, estimated to represent ca. 23.5% of the British breeding population (Stroud et al. 2001). Stroud et al. (2014) estimated that the six SPAs held about 2,863 pairs in counts made around 1999-2009, but data used for several of these colonies came from 1999-2000 so are rather out of date. Their estimate was that the SPA suite then held about 16.8% of the GB population. However, the most recent counts for these sites (Table 14.1) sum to only 1,826 pairs, with half of these being at Isles of Scilly SPA, so if the UK population is around 16,800 pairs the data suggest that the SPA suite now holds close to 11% of the population, with the single SPA in SW England being by far the largest contribution, due to very large declines in the colonies in north Scotland.



Figure 14.7. The SPA suite for breeding great black-backed gulls. These SPA populations are listed in Table 14.1.

Table 14.1. The UK SPA suite for breeding great black-backed gulls.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea							
Calf of Eday	Orkney	938 (1996)	1998	Declined 2006	675 100 281	2000 2004 2006	Stroud et al. 2014 Lewis et al. 2012 Lewis et al. 2012
Copinsay	Orkney	600	1994	Declined 2008	324 218	2008 2010	Lewis et al. 2012 SCM database
East Caithness Cliffs	N Scotland	850	1996	Declined 1999	175	1999	Seabird2000
Hoy	Orkney	570	2000	Maintained 2000	438 ca.60	2000 2011	Stroud et al. 2014 SMP database
West of Scotland							
North Rona & Sula Sgeir	N Scotland	733 (1986)	2001	Declined 2012	350 191	2009 2012	SMP database SMP database
SW and Channel							
Isles of Scilly	SW England	766	2001		901	2006	SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

14.10 BDMPS

UK waters can be divided into three BDMPS for great black-backed gull. It would be difficult to divide the North Sea region into separate northern and southern BDMPS populations because great black-backed gulls in the North Sea appear to be fairly mobile in the non-breeding period, changing distribution with movement of trawl fishery fishing effort, and because there have been no studies of great black-backed gulls using tracking methods, the details of movements of birds from particular sites are not known. In the UK North Sea BDMPS the population appears to be dominated by birds arriving from Barents Sea colonies in late summer and remaining until spring. There are probably about 910,000 birds in the area in the non-breeding season, with about 29,000 coming from the UK population and 63,000 from the Barents Sea. The area west of Scotland is quite distinct from the North Sea BDMPS because very few birds from the Barents Sea population enter the west of Scotland area, and few birds from North Sea colonies cross into west of Scotland. Similarly, few birds from west of Scotland colonies cross to the North Sea. In the West of Scotland BDMPS there are probably about 34,000 birds in the area in the non-breeding season, with about 25,000 from the UK population and 10,000 from the Barents Sea, Irish and Faroe populations. The southwest of Britain and Channel represents another distinct BDMPS for this species because birds in that area originate from local colonies in that area, together with rather small numbers of immatures from colonies further north in west of Scotland area, and very small numbers of birds from overseas (mostly Ireland). In the South-west and Channel BDMPS there are probably about 18,000 birds in the non-breeding season, with about 14,000 from the UK population and 4,000 from the Barents Sea, Irish and Faroe Populations (most of those coming from Irish colonies).

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 44 to 46.

Based on evidence reviewed in sections 14.5, 14.6 and 14.7, in the non-breeding season (September to March) in the UK North Sea, the BDMPS is estimated to include 100% of adults and 100% of immatures from colonies on the UK North Sea coast, 1% of adults and 10% of immatures from UK colonies in western Scotland, Northern Ireland, Wales and W England (Appendix A Table 44). The BDMPS is also estimated to include birds from two overseas populations; 30% of adults and 50% of immatures from the Barents Sea, 30% of adults and 30% of immatures from Faroe, but no birds from Ireland. These proportions result in an estimated BDMPS of 91,399 birds in the UK North Sea in the non-breeding season, 28,663 from the UK and 62,736 from overseas.

Based on evidence reviewed in sections 14.5, 14.6 and 14.7, in the non-breeding season (September to March) in the UK West of Scotland, the BDMPS is estimated to include no adults or immatures from colonies on the UK North Sea coast, 99% of adults and 80% of immatures from UK colonies in western Scotland, but none from Northern Ireland, Wales and W England (Appendix A Table 45). The BDMPS is also estimated to include birds from three overseas populations; 1% of adults and 8% of immatures from the Barents Sea, 10% of adults and 30% of immatures from Faroe, and 10% of adults and 20% of immatures from Ireland. These proportions result in an estimated BDMPS of 34,380 birds in UK West of Scotland in the non-breeding season, 24,703 from the UK and 9,677 from overseas.

Based on evidence reviewed in sections 14.5, 14.6 and 14.7, in the non-breeding season (September to March) in the UK South-west waters and Channel, the BDMPS is estimated to include no adults or immatures from colonies on the UK North Sea coast, no adults but 10% of immatures from UK colonies in western Scotland and Northern Ireland, 90% of adults and 70% of immatures from colonies in SW England (Appendix A Table 46). The BDMPS is also estimated to include birds from three overseas populations; no adults but 2% of immatures from the Barents Sea, no adults but 20% of immatures from Faroe, 10% of adults

and 30% of immature from Ireland. These proportions result in an estimated BDMPS of 17,742 birds in the UK SW waters and Channel in the non-breeding season, 13,663 from the UK and 4,079 from overseas.

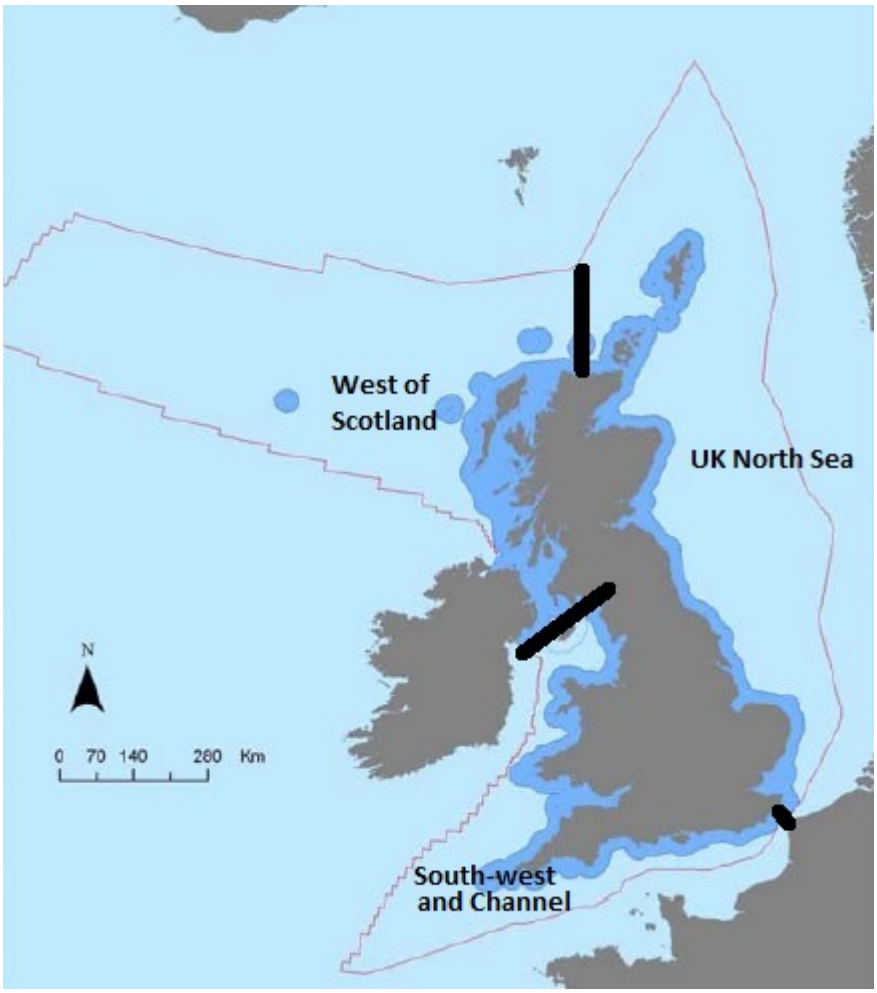


Figure 14.8. Three defined BDMPS spatial areas for great black-backed gull: 'UK North Sea', 'West of Scotland' and 'South-west and Channel'.

14.11 Proportions of UK SPA birds in each BDMPS

The UK North Sea BDMPS holds four of the six UK SPAs for breeding great black-backed gulls, but breeding numbers in these colonies have decreased dramatically. There are now probably no more than 700 pairs in total at these four sites combined, and possibly fewer than 600 given that no count data are available since 1999 for East Caithness Cliffs SPA or since 2006 for Calf of Eday SPA. The BDMPS of 91,399 birds in the UK North Sea is likely to contain only about 1,490 adults from UK SPA populations (Appendix A Table 44). So UK SPA breeding adults represent only about 2% of the BDMPS population in that area. The West of Scotland BDMPS holds only one SPA population, on North Rona and Sula Sgeir SPA. This contributes 378 adults to the non-breeding BDMPS, represent about 1% of the BDMPS total of birds (Appendix A Table 45). Ringing data suggest that very few birds from North Sea colonies (including Orkney and Shetland) move out of the North Sea into the West of Scotland region, so these populations appear to be fairly discrete, though it is less certain that birds from North Rona remain entirely in the West of Scotland rather than moving into the North Sea, as few birds have been ringed at North Rona. The UK South-west waters and Channel BDMPS contains one SPA population, Isles of Scilly SPA. There were 901 pairs there in 2006 and that population, in contrast to those in Scotland, appears to be increasing or at least stable (Table 14.1). The UK SPA breeding adults contributing to

that non-breeding season BDMPS (1,622 adults) represent about 9% of the BDMPS in UK South-west waters and Channel (Appendix A Table 46).

14.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Adult great black-backed gulls from UK colonies may remain very close to the colony throughout the year, while immatures tend to move south but not over very large distances. So the distribution of UK SPA birds within the BDMPS is likely to be aggregated in waters close to SPA colony sites. This may be especially the case in the West of Scotland BDMPS, with adult birds from North Rona mainly being close to North Rona, and in UK South-west waters and Channel with adult birds being around the Scillies all through the year. However, no detailed tracking studies have been carried out with great black-backed gulls, so the interpretation is based on ring recovery data and it would be useful to support that with work deploying geolocators on this species at major SPA colonies.

15. BLACK-LEGGED KITTIWAKE *Rissa tridactyla*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in autumn (August to December) (adults and immatures)	Numbers in UK waters in spring (January to April) (adults and immatures)
Overseas	4,020,000	1,017,320	567,136
UK	1,080,000	724,203	752,206
Total	5,100,000	1,741,523	1,319,342

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Autumn migration BDMPS (August to December)			
UK North Sea	829,937	397,808	432,129
UK Western waters plus Channel	911,586	619,512	292,074
Spring migration BDMPS (January to April)			
UK North Sea	627,816	238,424	389,392
UK Western waters plus Channel	691,526	328,712	362,814

Breeding adult kittiwakes have been equipped with geolocators in many different countries to investigate migrations and wintering areas, and that work has been summarised in a detailed paper by Frederiksen et al. (2012). However, it must be recognised that the geolocator study provides data for only a single winter, so that annual variation is not assessed, and provides data only for breeding adults, so that comparison with movements of immature birds cannot be made. There is other evidence indicating that individual breeding kittiwakes may differ in their migration behaviour from year to year depending on their breeding success, and that numbers of kittiwakes passing through UK waters vary strongly from year to year apparently in relation to weather conditions. Ring recovery data for kittiwakes are quite limited, and with a pelagic seabird tend to provide a biased indication of distribution. Geolocator data show rather different pattern from ring recovery data. In addition to this uncertainty about movement patterns, and evidence that these show high variability, there is also considerable uncertainty about very recent changes in kittiwake population sizes; several populations

appear to be in decline, but the extent and scale of decline are uncertain for most overseas populations. While breeding numbers at some UK SPA colonies have been counted since Seabird2000, some particularly large populations have not been counted since 2000 (e.g. East Caithness Cliffs where over 40,000 pairs nested in 1999). Many non-SPA colonies in the UK have not been counted recently. Changes in breeding numbers differ between Shetland (extreme decline), Orkney (decline in some colonies but perhaps not in others), southern Scotland (more stable numbers), and Wales (increases in some colonies but declines in others). Therefore, overall, numbers from UK in BDMPS are coded amber, and numbers from overseas are coded red, as are total numbers in BDMPS.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 47 to 50.

15.1 Breeding range and taxa

This Holarctic breeding species is usually split into two subspecies; *R. t. pollicaris* breeds in the North Pacific and does not normally reach the Atlantic. *R. t. tridactyla* breeds in the North Atlantic from Spain to the Arctic Ocean. Because *R. t. pollicaris* does not normally reach the Atlantic Ocean, this report focuses only on the nominate subspecies *R. t. tridactyla.* There is clinal variation in size, with birds from further north being larger (Barrett et al. 1985), but there does not seem to be much use of this variation to assess origins of individual birds.

15.2 Non-breeding component of the population

Kittiwakes start to breed when 4 years old (BTO Birdfacts). Coulson (2011) gives mean ages at first breeding of 3.97 years for males and 4.7 years for females at North Shields. Adult survival rate is 0.882 (BTO Birdfacts), juvenile survival 0.79 (BTO Birdfacts) and mean productivity is 0.672 chicks per pair (JNCC database, n=189 measurements), though this is strongly influenced by sandeel abundance near to the colony (Frederiksen et al. 2005). Coulson (2011) presents a table listing estimated adult survival rates for studies of kittiwakes breeding at North Shields, Marsden, Skomer, Brittany, Foula, Isle of May, Fair Isle, and colonies in north Norway and Alaska. Adult survival rate varied with period and colony, ranging from 0.8 to 0.93, indicating that this parameter is certainly not a constant for the species. To obtain a stable population, survival of immatures was adjusted to 0.68 for juveniles, and set at 0.76 for 1-year olds, 0.8 for 2-year olds, and 0.86 for 3-year olds. The model population comprised 53% adults, 18% juveniles and 29% older immatures. There are 0.88 immatures per adult.

15.3 Phenology

Breeding colonies in the UK are deserted in August, with modal departure in early August (Pennington et al. 2004; Forrester et al. 2007). Autumn dispersal/migration starts in July (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007) or August (Cramp et al. 1977-94). Peak autumn migration occurs in August-September in Shetland (Pennington et al. 2004), August-November in Scottish waters (Forrester et al. 2007), September-November throughout Europe (Cramp et al. 1977-94), but as late as October-November in Belgium (Vanermen et al. 2013). Variation in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) was erratic and not clearly indicative of autumn migration (Figure 15.1). Autumn migration is completed by December (Cramp et al. 1977-94; Pennington et al. 2004; Forrester et al. 2007).

Spring migration starts in January (Pennington et al. 2004) or January-February (Cramp et al. 1977-94; Forrester et al. 2007). Peak spring migration occurs in January-April in Belgium (Vanermen et al. 2013), in March-April generally in Europe (Cramp et al. 1977-94; Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites

(predominantly in south and east England) occurred in March (Figure 15.1). Spring migration is completed by May (Cramp et al. 1977-94; Forrester et al. 2007).

The first spring records of kittiwake in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly from January and the last records were predominantly in December. Peak autumn migration was reported in August-October in most years, and peak spring migration was reported in April in most years. Birds re-occupy colonies from February, with modal return in March (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007). Recent studies of kittiwakes have shown that corticosterone levels influence migratory and breeding behaviour. Experimentally increased levels of corticosterone caused female kittiwakes to migrate away from the breeding colony earlier and to spend longer on the wintering grounds (Schultner et al. 2014), while in years with poor food availability, corticosterone levels increased in kittiwakes, birds bred later and made longer foraging trips travelling further from the colony in the pre-breeding period (Goutte et al. 2014). Although demonstrated in kittiwakes, these patterns seem likely to apply in all seabirds.

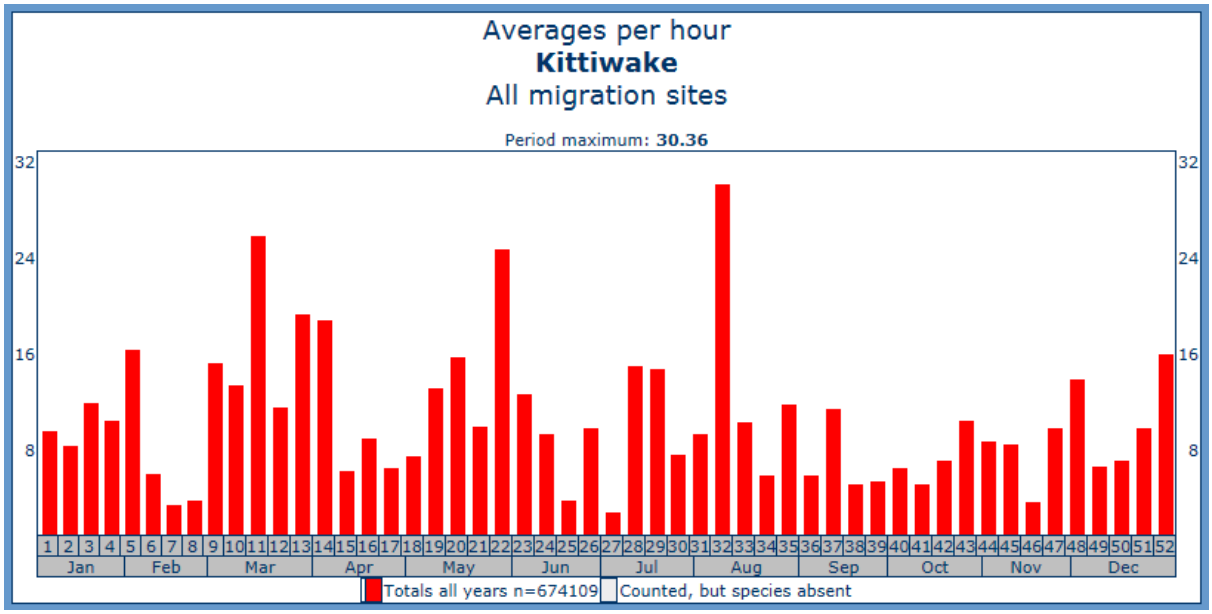


Figure 15.1. Average numbers of kittiwakes counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-September, non-breeding season October-April. However, from the data reviewed above, a more appropriate definition would be breeding season March-August, non-breeding season September-February.

15.4 Defined seasons:

- UK Breeding season — March-August
  - Migration-free breeding season — May-July
- Non-breeding season — September-February
  - Post-breeding migration in UK waters — August-December (**autumn BDMPS**)
  - Return migration through UK waters — January-April (**spring BDMPS**)

Apart from the breeding season, two seasonal BDMPS periods are considered to be appropriate for black-legged kittiwake:

'Autumn' (post-breeding) migration BDMPS (August-December); and

'Spring' (pre-breeding) migration BDMPS (January-April).

15.5 Movements of birds from the UK population

In the UK, kittiwake chicks disperse rapidly from colonies, leaving the area about 10 days on average after their first flight (Coulson 2011). Rapid dispersal is consistent with the fact that chicks are not fed by their parents after departing, so can depart without constraint (Coulson 2011). After initial dispersal which can be in any direction with birds congregating where food is available, subsequent autumn migration takes some young birds west across the Atlantic and others south towards Iberia (Wernham et al. 2002). Kittiwakes in winter may be distributed all across the North Atlantic and North Sea, regularly as far south as about 40^{o}N, but with a few birds even crossing into the southern hemisphere (Coulson 2011). The main spread southwards occurs in early October, birds reaching their southernmost distribution in December-January (Coulson 2011). Ring recovery data show that in spring, young birds may move north, with birds on the west side of the Atlantic visiting seas around Greenland, and birds on the east side possibly moving north but not as far as their breeding colony (Coulson 2011). However, in their first summer and in subsequent summers, kittiwakes vacate the open ocean areas they occupy in winter, and move into shallow continental shelf waters, and may rest on shores though generally away from colonies (Coulson 2011). Immature birds follow a similar pattern to juveniles (although a few two year olds do return to the colony in summer if only briefly), and then tend to return towards breeding colonies in their third summer, though even at that age some may remain in the west Atlantic (Wernham et al. 2002; Coulson 2011). Adults depart from colonies in the northern part of the UK rather rapidly in late July or early August, apparently at least in part in response to sandeels becoming unavailable towards the end of the summer. Further south, adults may linger near colonies for longer. Some adults cross the Atlantic to winter off Newfoundland, but there are far more recoveries of adult kittiwakes in the east Atlantic (Wernham et al. 2002). Ring recoveries indicate that British kittiwakes tend to winter further south than those from colonies in the far north of Europe, so populations only show partial overlap outside of the breeding season (Wernham et al. 2002; Coulson 2011).

15.6 Movements of birds from overseas into UK waters

Over 100 foreign-ringed kittiwakes have been recovered in the British Isles, mostly in autumn and winter. Those birds originated mainly from Norway, Russia, France, and the Channel Islands. Only small numbers of recoveries originated from Iceland, Faroe, Denmark, Sweden, Germany and Greenland. Deployment of geolocators on breeding kittiwakes at many colonies in Svalbard, Barents Sea, Norwegian Sea, North Sea, Celtic-Biscay Shelf, Faroe, Iceland, Greenland and Canada (Frederiksen et al. 2012) has provided more detailed information on the migrations and wintering areas of 236 adult breeding status kittiwakes from different North Atlantic populations. Those data are largely consistent with the ring recovery data, but tend to more strongly emphasise the tendency for birds to cross to the west side of the Atlantic, strongly suggesting that ring recovery data under-represent trans-Atlantic movements. Geolocation data must be considered with some caution, as they are not available from all kittiwake populations in the North Atlantic, they represent only birds of breeding adult status, and data were collected in only two years (2008-09 and 2009-10), so may not be typical of kittiwake migration behaviour in other years. Nevertheless, the geolocation data provide detailed information on the movements of a large sample of birds from many different regions and colonies. Details of this study can be accessed at http://www.hav.fo/PDF/Ritgerdir/2011/Kittiwake_paper_Bergur.pdf. Most tracked birds moved to the west Atlantic to winter between Newfoundland and the mid-Atlantic ridge.

Some wintered in the North Sea and west of the British Isles, and those birds mostly came from colonies in the British Isles or from colonies in the Barents Sea. No birds from colonies in west Atlantic wintered in Europe. There was considerable overlap in winter distributions of birds from different colonies, although colonies closer together showed greatest overlap in distribution, so there was some spatial structuring. Overall, about 80% of the 4.5 million breeding adult kittiwakes in the Atlantic were estimated to winter west of the mid-Atlantic ridge, with only birds from British Isles and France remaining predominantly on the European side. Many equipped birds remained near to their breeding site throughout August, but some moved to post-breeding aggregations in the Barents Sea, the Denmark Strait, and the Labrador Sea. In November most birds had reached wintering areas mostly south of 62^{o}N, but some birds remained in the Norwegian Sea. In December, most birds were in the west Atlantic, but with substantial numbers in the North Sea and west of the British Isles. By January, some birds were returning towards breeding sites. Most birds were back at breeding sites by April, but some high-Arctic breeders remained offshore in the Barents Sea or Davis Strait or off Newfoundland. Frederiksen et al. (2012) present electronic supplementary material to their paper indicating estimates that 255,261 adult kittiwakes were present in the entire North Sea (not just the UK portion) in December 2009, with 102,671 of these from Barents Sea colonies, 114,195 from North Sea colonies, 24,071 from Norwegian Sea colonies, and 14,324 from Celtic Shelf colonies. In the Celtic-Biscay Shelf area they estimate that there were 345,288 adult kittiwakes in December 2009, with 189,934 from Celtic-Biscay shelf colonies, 116,027 from Barents Sea colonies, 39,180 from North Sea colonies, and 147 from Norwegian Sea colonies. While these detailed data are extremely valuable, it must be remembered that these only apply to adult kittiwakes and not immatures, and only apply to a single winter, so it is uncertain whether these are typical or not. Kittiwakes may return to breeding colonies from mid-February in the UK, though not until April in the Arctic (Coulson 2011). To complicate this picture further, Bogdanova et al. (2011) found that unsuccessful breeding kittiwakes from the Isle of May colony were more likely than successful breeders to migrate to the west Atlantic area. Males and females may also differ in migratory behaviour although this is less certain (Bogdanova et al. 2011). The difference in migration behaviour of successful and failed breeders could indicate a time constraint to the migration to the west Atlantic, as birds that fail in their breeding attempt tend to leave the colony earlier in the summer than successful breeders. Since breeding success was very poor at many kittiwake colonies in the eastern Atlantic in the two years when geolocators were deployed, it is possible that the proportion of adults migrating to the west Atlantic was higher than in other years.

15.7 Numbers in UK waters

Although clearly an abundant seabird, kittiwake numbers in UK waters during migration and winter are not well known, and apparently vary considerably, perhaps in relation to food supply and weather conditions. ESAS data suggest a total of around 1,500,000 birds in the North Sea in autumn migration period, with the majority of these birds in the NW North Sea (up to 700,000 birds) and off the English north-east coast (up to 200,000 birds) (Camphuysen et al. 1995), with at sea densities of around 4 birds per km^2. From surveys in 2007 and 2008, Fauchald and Tveraa (2009) reported mean densities at sea of 24-60 birds per km^2 in the Norwegian Sea in spring/summer, and 15-54 birds per km^2 in the Barents Sea in autumn, so densities in the North Sea are not high when compared with some other regions. Breeding numbers in Iceland declined by 17% from 630,000 pairs in 1983-86 to 523,000 pairs in 2005-08 (Gardarsson 2006), but apparently Icelandic kittiwakes do not visit UK waters. However, breeding numbers of kittiwakes have apparently been declining throughout most of the North Atlantic over recent years, so numbers are almost certainly lower in most countries than they were in the period that informed total population estimates in Stroud et al. (2001) and Mitchell et al. (2004). Forrester et al. (2007) suggest that there may be about 10,000 birds in Scottish inshore waters in winter, but give no estimate for numbers in offshore waters. Frederiksen et al. (2012) present electronic supplementary

material to their paper estimating that 255,261 adult kittiwakes were present in the entire North Sea (not just the UK portion) in December 2009. This would suggest that taking immatures into account (and the fact that a higher proportion of immatures move across to the west Atlantic) there would probably be about 200,000 kittiwakes in UK North Sea waters in winter. Camphuysen et al. (1995) estimated that there were about 300,000 to 1,100,000 kittiwakes in the (entire) North Sea in February based on surveys in 1993 and 1994 and ESAS data, with the largest proportion of these in UK sectors of the North Sea. Densities of kittiwakes in inshore waters west of the UK in winter are very low indeed; close to zero. Offshore, densities in winter are low, but highly variable as occasional large numbers pass through UK waters in winter, apparently in response to weather more than to food. During autumn, large numbers disperse from UK colonies out of UK waters, returning in spring. Birds from populations further north pass through western UK waters in autumn, and to a lesser extent in spring, but the absolute numbers involved are very uncertain, despite the detailed tracking reported by Frederiksen et al. (2012).

15.8 Biogeographic population and relevant smaller units (BDMPS)

Stroud et al. (2001) defined the biogeographic breeding population as that of the North Atlantic population, comprising 3,170,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 2,500,000-3,000,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 8,400,000 individuals. Counts in the UK suggest a breeding population of around 288,500 pairs (Appendix Table 47). Summing populations with connectivity to UK waters gives an estimated total of about 1,270,000 pairs (Figure 15.2); the huge size of populations in the Barents Sea is a major part of this total. Numbers in the Barents Sea have apparently not declined as much as numbers in the UK and probably in Faroe, but there is low confidence in the exact numbers at Barents Sea colonies and how much these have changed (Frederiksen 2010, Frederiksen et al. 2012). Numbers in Norway have declined too, but there is some uncertainty about how much and how this pattern varies regionally (Barrett et al. 2006). The biogeographic population with connectivity to UK waters is therefore a total of about 5.1 million birds, 1.08 million from UK and 4.02 million from overseas. However, only very small proportions of these overseas populations are found in UK waters during migration seasons (autumn; August to December, and spring; January to April). The estimated total numbers in UK waters in autumn are 1,740,000 birds (720,000 from UK, 1,020,000 from overseas) and 1,320,000 birds in spring (750,000 from UK, 570,000 from overseas).

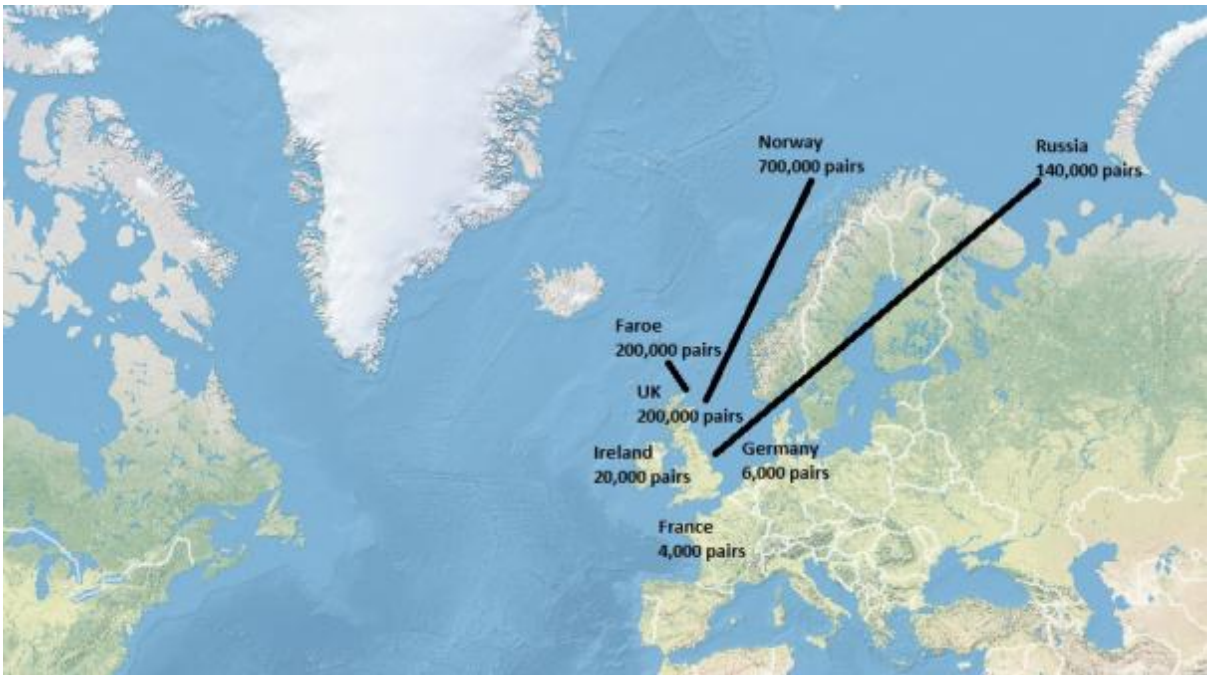


Figure 15.2. Breeding population origins of kittiwakes in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

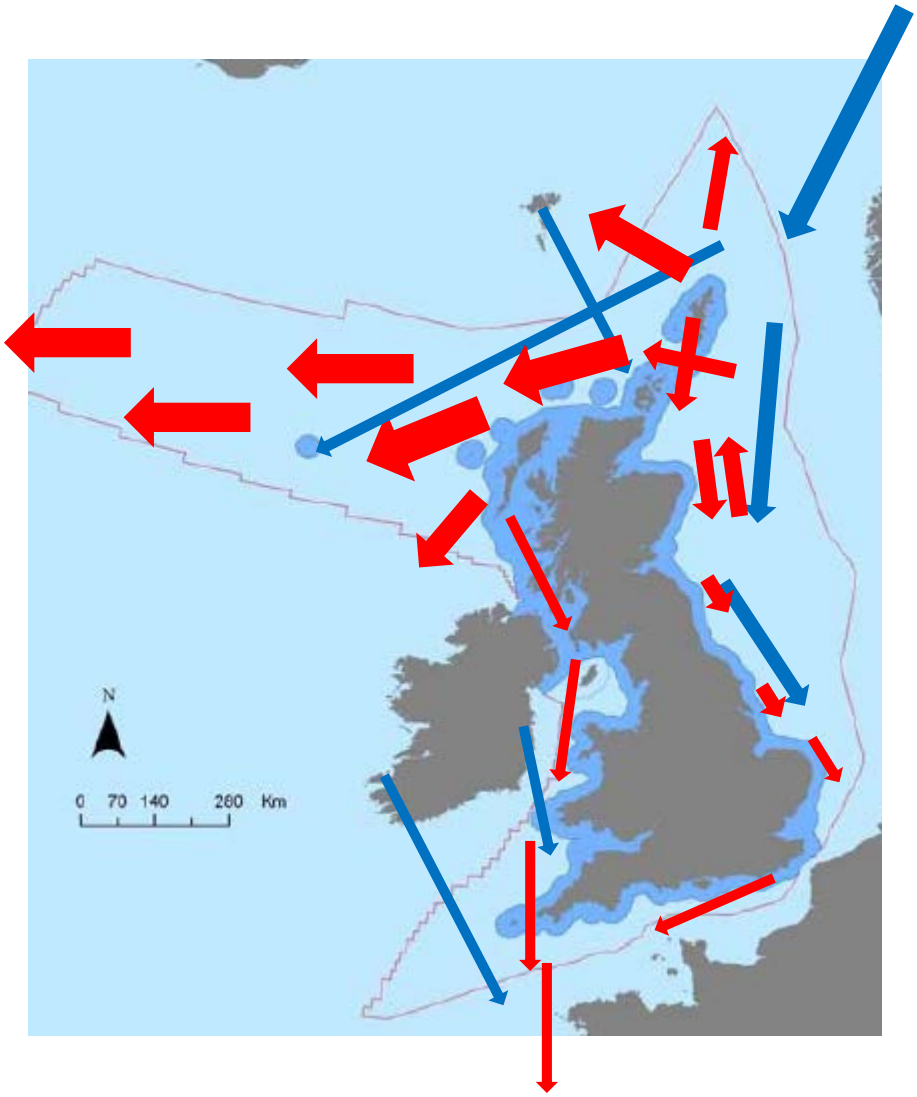


Figure 15.3. Main movements of kittiwakes from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

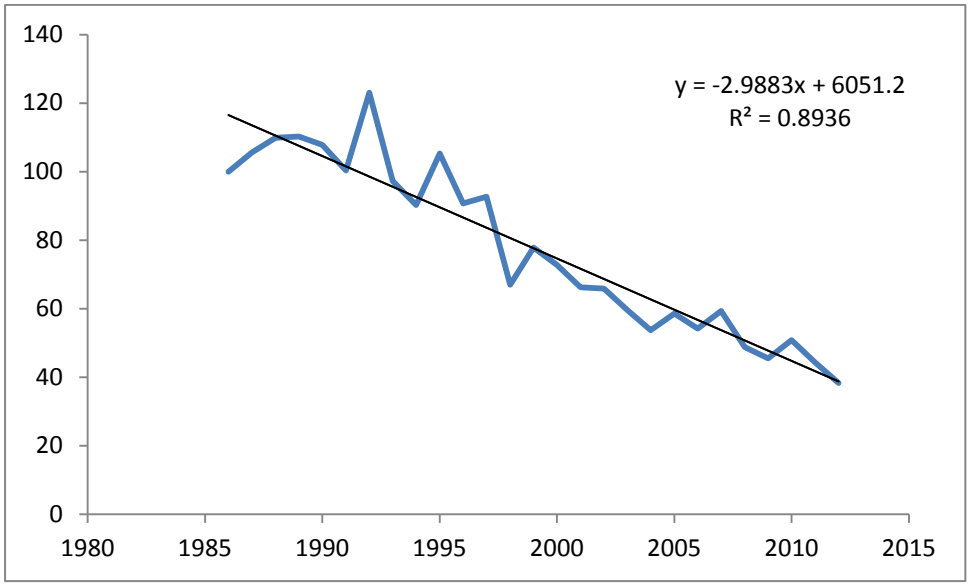


Figure 15.4. Trend in the kittiwake breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

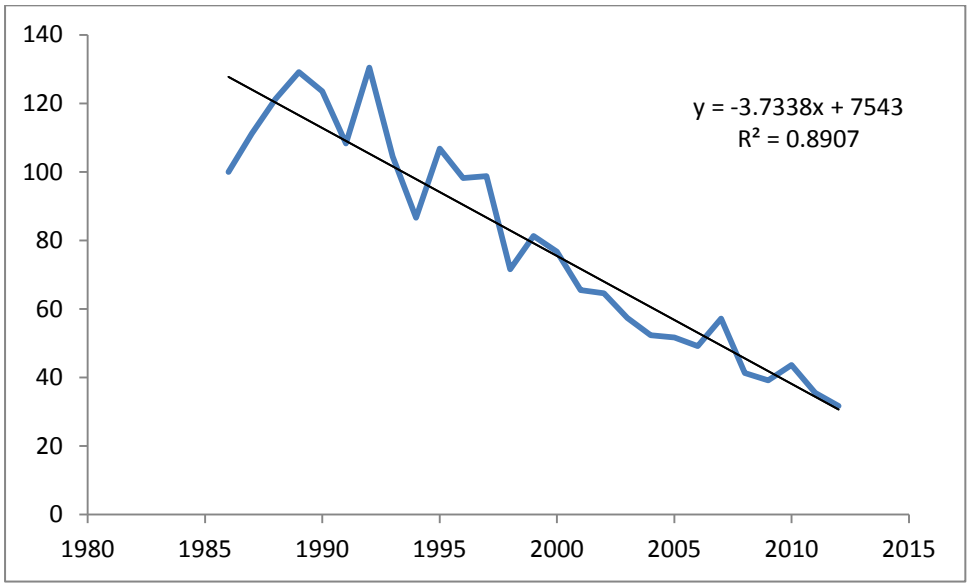


Figure 15.5. Trend in the kittiwake breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

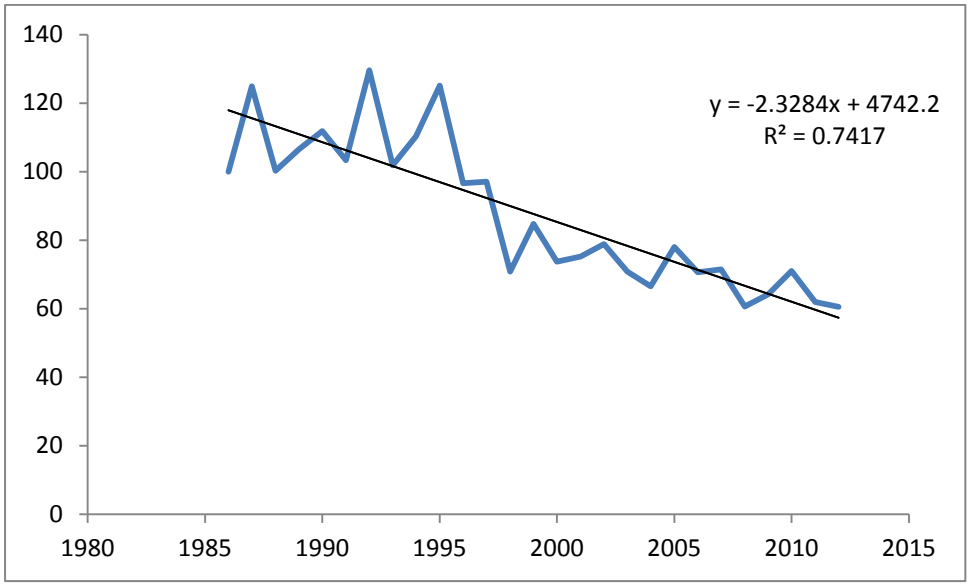


Figure 15.6. Trend in the kittiwake breeding population index in England from 1986-2012. Data from JNCC seabird population monitoring database.

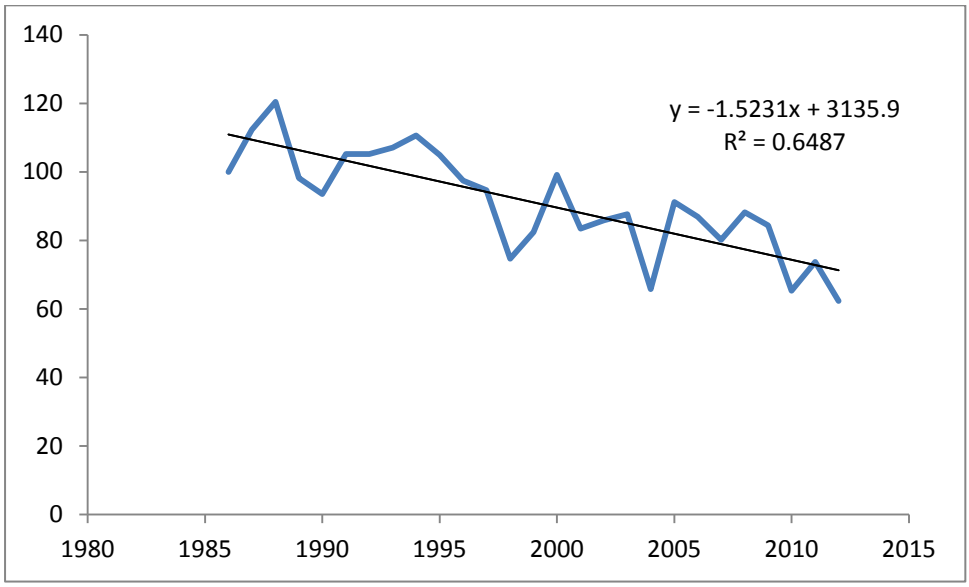


Figure 15.7. Trend in the kittiwake breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

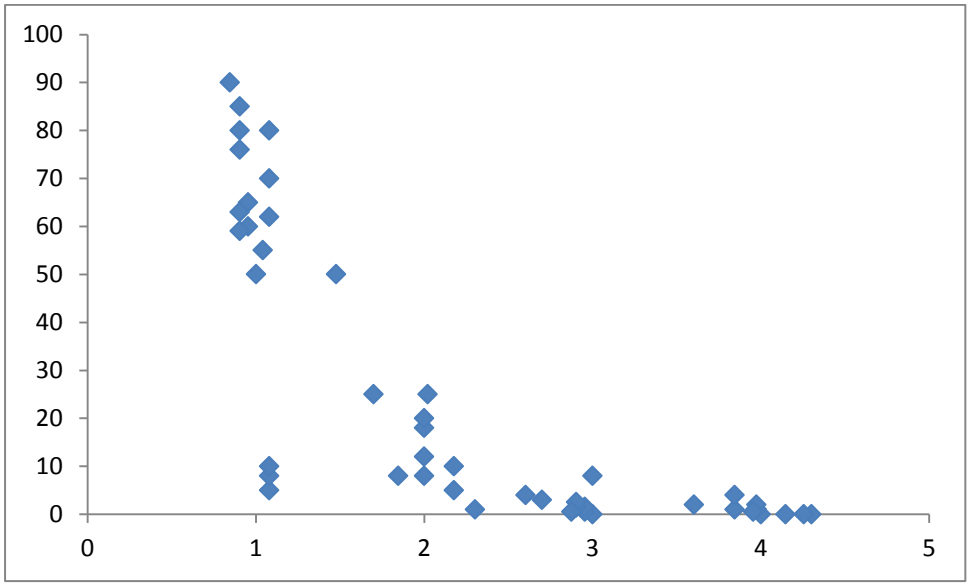


Figure 15.8. Percentage increase in kittiwake colony size (number of nests) at 46 colonies in the UK between surveys in 1959 and 1969 (from Coulson 2011), showing the density-dependent relationship between colony size and growth rate during this period of rapid population growth. Colony size is on a log scale. The same sort of density-dependent relationship between growth rate and colony size has been shown for other time periods so this graph is simply one example of this general phenomenon.

15.9 Proportion of UK population in UK breeding SPAs

The 33 SPAs with breeding kittiwakes as a feature together held 390,597 pairs at designation, estimated to represent ca. 78% of the British breeding population (Stroud et al. 2001). However, based on census data for 1999-2011, Stroud et al. (2014) estimated that this suite held 56.5% of the GB population, as many of the largest colonies have declined even more than the population as a whole. Since a number of the colony size estimates used by Stroud et al. (2014) were from 1999 or 2000, so are very likely to be considerable overestimates of numbers in those colonies now, the true percentage of the population in the SPA suite for breeding kittiwakes is likely to be slightly lower than the estimate in Stroud et al. (2014), perhaps around 55% now.

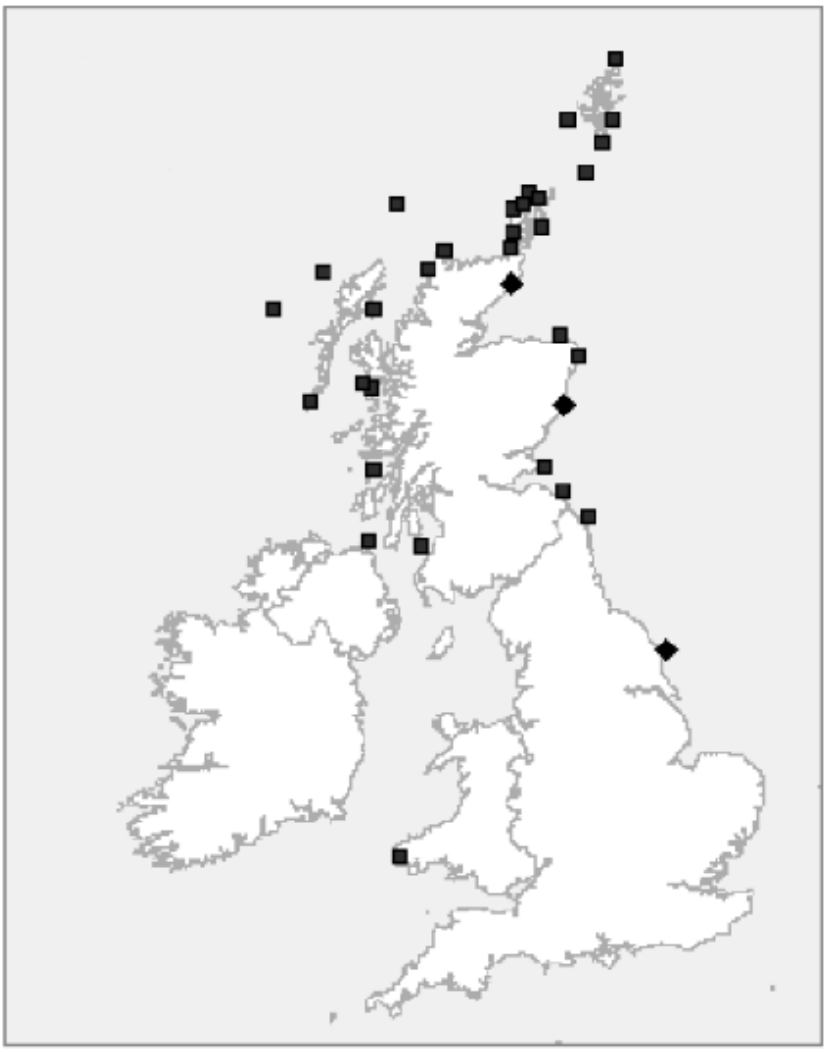


Figure 15.9. UK SPA suite for breeding kittiwakes. These SPA populations are listed in Table 15.1.

Table 15.1. The UK SPA suite for breeding kittiwakes.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent counts	Year	Reference
UK North Sea							
Hermaness, Saxavord & Valla	Shetland	1,710	1994	Declined 2009	710 624 490 391	1999 2002 2005 2009	SCM database SCM database SCM database SCM database
Foula	Shetland	3,840	1995	Declined 2007	997 509 582 480 378 327	2007 2009 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SCM database Gear 2012 Gear 2013
Noss	Shetland	4,270	1996	Declined 2005	2,395 1,427 507	2000 2005 2010	Seabird2000 Lewis et al. 2012 Lewis et al. 2012
Sumburgh Head	Shetland	1,366 (1994)	1996	Declined 2007	506 500 549 210	2007 2009 2010 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SCM database

Fair Isle	Shetland	9,660	1994	Declined 2008	2,688 1,438 1,225 771	2008 2011 2012 2013	SCM database SCM database FIBO Report SCM database
West Westray	Orkney	24,000	1996	Declined 2007	33,281 12,055	1999 2007	Seabird2000 Lewis et al. 2012
Calf of Eday	Orkney	1,717	1998	No change 2006	765 747	2002 2006	Lewis et al. 2012 Lewis et al. 2012
Marwick Head	Orkney	7,110	1994	Declined 2006	3,860 2,185 2,018 1,134 526	2003 2006 2009 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SCM database SCM database
Rousay	Orkney	4,900	2000	Declined 2009	2,713 1,764	1999 2009	Seabird2000 Lewis et al. 2012
Copinsay	Orkney	3,610	1994	Declined 2008	3,552 666	2008 2012	Lewis et al. 2012 SCM database
Hoy	Orkney	3,000	2000	Declined 2007	781 397	1999 2007	Seabird2000 Lewis et al 2012
North Caithness Cliffs	N Scotland	15,650	1996	Declined 2000	10,150	2000	Seabird2000
East Caithness Cliffs	N Scotland	31,930 (1986)	1996	Maintained 1999	40,410	1999	Seabird2000
Troup, Pennan and Lion's Heads	NE Scotland	31,660 (1995)	1997	No change 2007	18,482 15,570 17,171 14,896	2001 2004 2007 2007	Seabird2000 Lewis et al. 2012 Lewis et al. 2012 SCM database
Buchan Ness to Collieston Coast	NE Scotland	30,452	1998	No change 2007	13,330 14,133 12,542	2004 2007 2007	SCM database Lewis et al. 2012 SCM database
Fowlsheugh	NE Scotland	34,870	1992	Maintained 1999	11,140 9,454 9,337	2006 2009 2012	Lewis et al. 2012 Lewis et al. 2012 SCM database
Forth Islands	E Scotland	8,400 (1985) Or 9,380 (Stroud et al. 2001)	1990	Declined 2007	5,164 3,884 3,766 3,100	2007 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 SCM database SCM database
St Abb's Head to Fast Castle	E Scotland	19,600	1997	Declined 2008	15,430 c.5,000 4,314 3,403	2000 2011 2012 2013	Seabird2000 Lewis et al. 2012 SCM database SCM database
Farne Islands	NE England	6,236	1985		4,275 3,699 4,768 3,976 4,241 3,443	2008 2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database SCM database
Flamborough Head & Bempton (to be subsumed into Flamborough and Filey Coast SPA)	E England	83,370 (1987)	1993		42,692 37,617	2000 2008	SCM database SCM database

Flamborough and Filey Coast	E England	44,520 (2008-2011)	Not yet		42,692 37,617	2000 2008	SCM database SCM database
UK Western waters & Channel							
Cape Wrath	NW Scotland	9,660	1996	Maintained 2000	10,344	2000	Seabird2000
North Rona and Sula Sgeir	N Scotland	5,040 (1986)	2001	Declined 2012	4,119 1,253	1998 2012	Lewis et al. 2012 SCM database
Handa	NW Scotland	7,420	1990	Declined 1999	7,013 5,985 4,466 1,872	1999 2005 2009 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SCM database
St Kilda	Western Isles	7,800 (1987)	1992	Maintained 2000	4,268 1,516 957	1999 2006 2008	Seabird2000 Lewis et al. 2012 Lewis et al. 2012
Flannan Isles	Western Isles	2,800 (1988)	1992	Declined 2013	1,392	1998	Seabird2000
Shiant Isles	Western Isles	1,850	1992	Maintained 1999	2,006 549	1999 2008	Seabird2000 Lewis et al. 2012
Canna and Sanday	Inner Hebs	1,193	1998	Maintained 2001	960 1,002 1,083 820	2010 2011 2012 2013	SCM database SCM database SCM database SCM database
Rum	Inner Hebs	1,500	1982	No change 2006	788	2000	Seabird2000
Mingulay and Berneray	Western Isles	8,610 (1985)	1994	Declined 2009	5,511 4,974 2,228	1998 2003 2009	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012
North Colonsay & Western Cliffs	W Scotland	4,512	1997	Maintained 2008	5,563	2000	Seabird2000
Ailsa Craig	W Scotland	3,100 (1987)	1990	Declined 2003	1,675 200 428 489	2001 2008 2009 2013	SCM database SCM database Lewis et al. 2012 SCM database
Rathlin Island	N Ireland	6,822 (1985)	1999		9,917 9,896 7,922	1999 2007 2011	SCM database SCM database SCM database
Skomer and Skokholm	Wales	1,959 Stroud et al. 2001)	1982		2,282 2,046 1,922 1,837 1,594 1,045	2008 2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database SCM database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

15.10 BDMPS

The UK waters can be divided into two spatially distinct BDMPS. Most birds from UK North Sea colonies are members of the UK North Sea BDMPS, whereas few birds from western colonies enter the North Sea. Conversely, although some birds from UK North Sea colonies enter UK western waters plus Channel, these are a minority from those populations whereas most birds from colonies in western waters contribute to the UK western waters plus Channel BDMPS. UK North Sea holds about 830,000 birds during autumn migration (August to December), and 630,000 in spring migration (January to April). It seems that slightly more than half of these birds are from the UK population. UK western waters plus Channel BDMPS holds about 910,000 birds during autumn migration, and 690,000 in spring

migration. It should be recognised, however, that although kittiwake numbers are undoubtedly large in both these populations, numbers are not known with confidence, and appear to be highly variable depending on weather patterns, and possibly also on food supply.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 47 to 50.

Based on evidence reviewed in sections 15.5, 15.6 and 15.7, the UK North Sea autumn migration BDMPS is estimated to contain 60% of adults and 40% of immatures from colonies in the UK North Sea, 1% of adults and 5% of immatures from colonies in UK western waters, 10% of adults and immatures from Russia, Norway, Faroe and Germany, 5% of adults and immatures from France and Ireland. This results in an estimated BDMPS population of 829,937 birds in autumn, 432,129 from UK and 397,808 from overseas (Appendix A Table 47).

Based on evidence reviewed in sections 15.5, 15.6 and 15.7, the UK western waters plus Channel autumn migration BDMPS is estimated to contain 20% of adults and 20% of immatures from colonies in the UK North Sea, 60% of adults and 40% of immatures from colonies in UK western waters, 10% of adults and immatures from Russia, 15% of adults and immatures from Norway, 20% of adults and immatures from Faroe and 5% of adults and immatures from Germany, 10% of adults and immatures from France, and 30% of adults and 20% of immatures from Ireland. This results in an estimated BDMPS population of 911,586 birds in autumn, 292,074 from UK and 619,512 from overseas (Appendix A Table 48).

Based on evidence reviewed in sections 15.5, 15.6 and 15.7, the UK North Sea spring migration BDMPS is estimated to contain 60% of adults and 30% of immatures from colonies in the UK North Sea, 1% of adults and 2% of immatures from colonies in UK western waters, 5% of adults and 7% of immatures from Russia, Norway, and Faroe, 15% of adults and 25% of immatures from Germany, 5% of adults and 10% of immatures from France, and 1% of adults and immatures from Ireland. This results in an estimated BDMPS population of 627,816 birds in spring, 389,392 from UK and 238,424 from overseas (Appendix A Table 49).

Based on evidence reviewed in sections 15.5, 15.6 and 15.7, the UK western waters plus Channel spring migration BDMPS is estimated to contain 30% of adults and 20% of immatures from colonies in the UK North Sea, 80% of adults and 40% of immatures from colonies in UK western waters, 5% of adults and 10% of immatures from Russia and Norway, 10% of adults and immatures from Faroe, 5% of adults and immatures from Germany, 10% of adults and immatures from France, 30% of adults and 20% of immatures from Ireland. This results in an estimated BDMPS population of 691,526 birds in spring, 362,814 from UK and 328,712 from overseas (Appendix A Table 50).

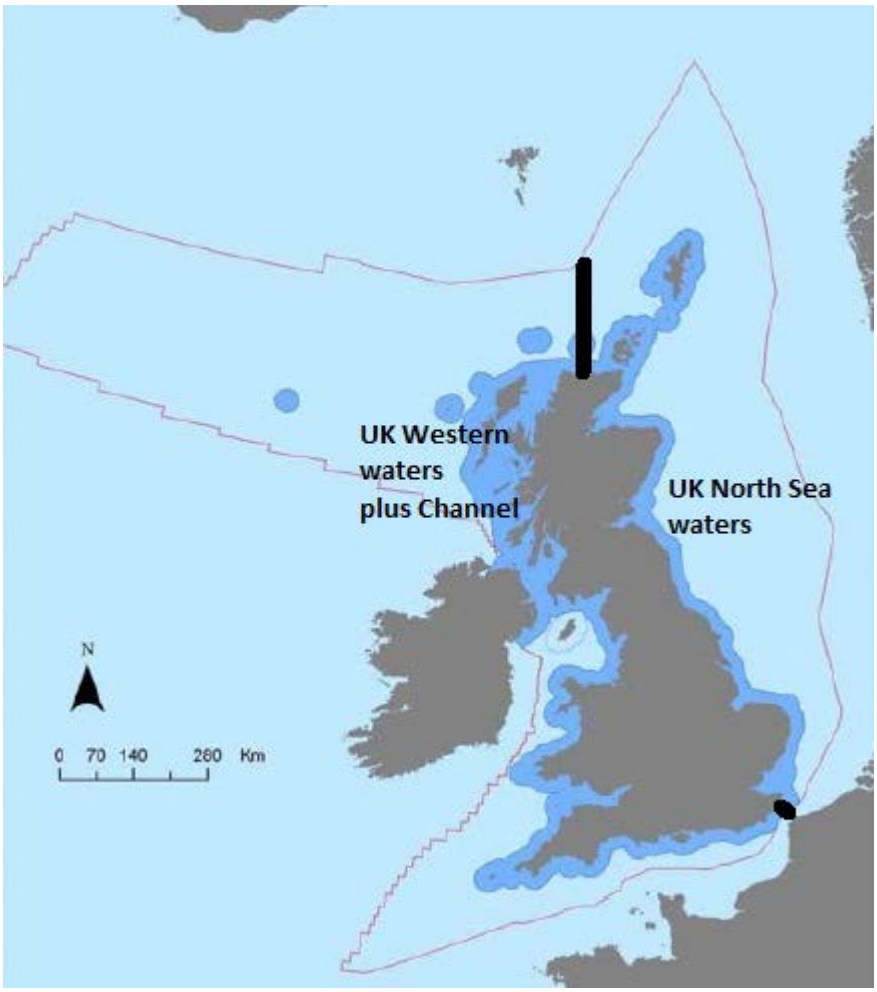


Figure 15.10. Two defined BDMPS spatial areas for kittiwake: 'UK North Sea waters' and 'UK Western waters plus Channel'.

15.11 Proportions of UK breeding SPA birds in BDMPS

The proportion of birds in each BDMPS that are adults from UK SPA populations can be estimated directly from Appendix A Tables 47 to 50. For example, the UK North Sea autumn migration season BDMPS comprises 829,937 birds in total, of which 184,615 are adults from UK SPA populations, giving an estimate of 22% being adults from UK SPAs.

15.12 Spatial distribution of UK breeding SPA birds across the BDMPS

The SPAs for breeding kittiwakes in UK North Sea BDMPS and UK Western waters plus Channel BDMPS are well distributed through the broad breeding range of the species in those areas. In the South-west and Channel area there is only one SPA population, in south Wales, so the distribution of SPA birds could be patchy, but since kittiwakes disperse very widely it is likely that in all areas they are very thoroughly mixed through the broader UK population and with birds from overseas.

16. SANDWICH TERN *Thalasseus sandvicensis*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (July-September and March-May)
Overseas	107,000	13,560
UK	41,000	35,252
Total	148,000	48,812

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Migration seasons BDMPS (July-September and March-May)			
UK North Sea and Channel	38,051	10,090	27,961
UK Western waters	10,761	3,470	7,291

Sandwich tern numbers in UK SPA colonies are almost all monitored frequently. However, numbers in UK colonies that are not SPA populations are less well monitored, and do represent a substantial proportion of the UK total. Sandwich tern migrations have not been studied by geolocator deployment, and ringing recoveries from the migration period in UK waters are very limited. So understanding of details of Sandwich tern movements are relatively poor, especially to the extent that birds from overseas populations are concerned. While ring recoveries show that some birds from overseas pass through UK waters, the proportions of those populations doing so are very uncertain since ring recovery data are subject to considerable potential bias. Therefore, numbers of overseas birds and total numbers in the BDMPS are classed as red, whereas numbers from the UK population are classed amber.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 51 and 52.

16.1 Breeding range and taxa

Sandwich tern has a Holarctic breeding distribution in warm temperate latitudes. There are three subspecies, but only nominate *T. s. sandvicensis* occurs within British waters. There is no evidence that biometrics would allow origins of individuals to be identified. Most populations breed south of the UK. There are moderate numbers in Denmark and Germany, but few in Norway or Sweden.

16.2 Non-breeding component of the population

Sandwich terns start to breed when 3 years old (BTO Birdfacts). Adult survival rate is 0.898 (BTO Birdfacts), juvenile survival 0.358 (BTO Birdfacts) and mean productivity is 0.656 chicks per pair (JNCC database, n=174 measurements). To obtain a stable population, survival of immatures was adjusted to 0.55 for juveniles, 0.7 for 1-year olds, and 0.8 for 2-year olds. The model population comprised 61% adults, 20% juveniles and 19% older immatures. There are 0.63 immatures per adult.

16.3 Phenology

Breeding colonies in the UK are deserted by late September (Brown and Grice 2005), with modal departure in August (Pennington et al. 2004; Forrester et al. 2007). Autumn dispersal/migration starts in July (Cramp et al. 1977-94; Pennington et al. 2004; Forrester et al. 2007) or August (Wernham et al. 2002). Peak autumn migration occurs in August in Shetland (Pennington et al. 2004), and Scotland (Forrester et al. 2007), July-September in Belgium (Vanermen et al. 2013) or September throughout Europe (Cramp et al. 1977-94; Wernham et al. 2002). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred from July to September (Figure 16.1). Autumn migration is completed in UK waters by October (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007) or early November throughout the geographical range (Cramp et al. 1977-94).

Spring migration starts in late February in the winter quarters (Cramp et al. 1977-94) and in March in UK waters (Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Peak spring migration occurs in March-April in Belgium (Vanermen et al. 2013) and in English waters (Brown and Grice 2005), in April (Cramp et al. 1977-94; Wernham et al. 2002) in April-May in Scottish waters (Forrester et al. 2007) and in June in Shetland (Pennington et al. 2004). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in April (Figure 16.1). Spring migration is completed in May (Cramp et al. 1977-94; Wernham et al. 2002), June (Forrester et al. 2007) or July in Shetland (Pennington et al. 2004).

The first spring records of Sandwich tern in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 2 March to 26 April but predominantly in late March, and the last records were from 28 June to 31 December but mostly in October. Peak autumn migration was reported in August-September in most years, and peak spring migration was reported in April or May in most years. Birds re-occupy colonies from March, with modal return in April (Pennington et al. 2004; Forrester et al. 2007).

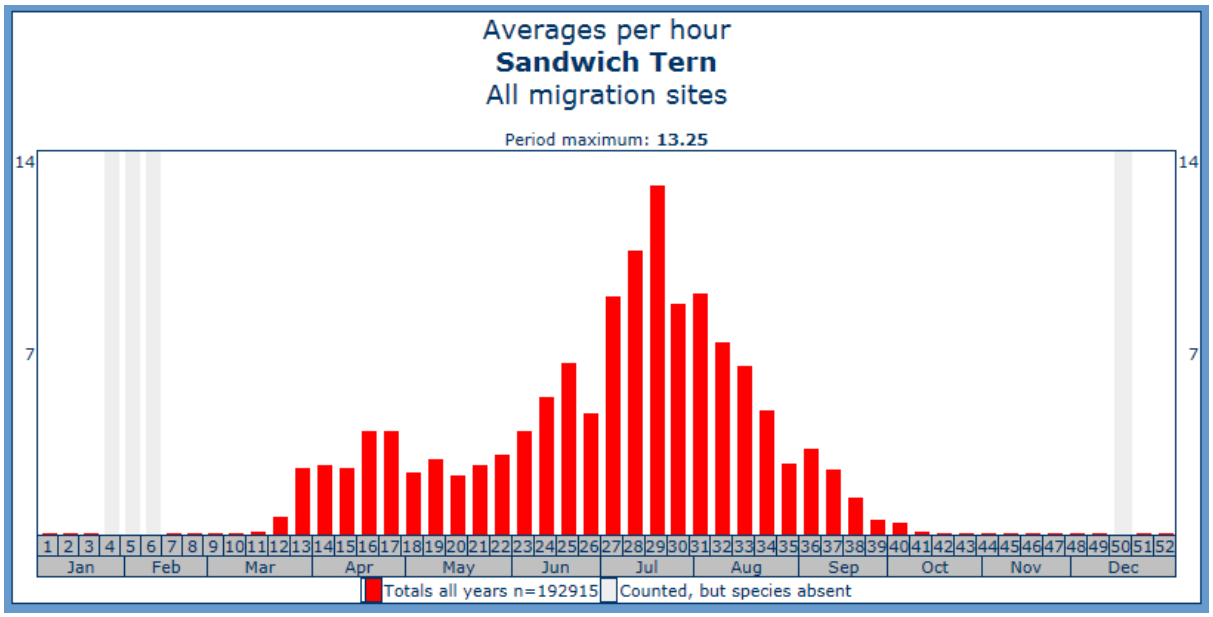


Figure 16.1. Average numbers of Sandwich terns counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season birds predominantly absent from UK waters. However, from the data reviewed above, a more appropriate definition would be breeding season April-August, non-breeding season September-March.

16.4 Defined seasons:

- UK Breeding season — April-August
- Post-breeding migration in UK waters — July-September (**migration BDMPS**)
- non-breeding season — September-March
- Return migration through UK waters — March-May (**migration BDMPS**)
- Migration-free breeding season — June
- Migration-free winter season — October-February

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for Sandwich tern:

Migration periods BDMPS (July-September, and March-May).

16.5 Movements of birds from the UK population

Sandwich terns breeding in the UK are mainly concentrated in a small number of colonies, with high concentrations in Norfolk, and Northumberland. Breeding adults may abandon colonies where habitat change, predation or flooding impact on productivity, and may move considerable distances to recruit into another colony, so European populations represent a large meta-population (Møller 1981). Birds begin to disperse from colonies in late June and many fledglings may cross the North Sea between continental and UK colonies in July-August (Wernham et al. 2002). Sandwich tern fledglings remain dependent on their parents for food for some weeks after fledging, so move as family parties rather than as independent individuals (Meissner and Krupa 2007). Birds move quite rapidly southwards to wintering areas from west Africa to southern Africa, so that very few remain in UK waters after September (Wernham et al. 2002), although there are small numbers seen as late as November on English coasts (Balmer et al. 2013).

16.6 Movements of birds from overseas into UK waters

Ring recoveries show movements of Sandwich terns from populations in Ireland, Denmark, Germany, The Netherlands and Belgium moving through UK waters. For example, birds caught at Teesmouth in late summer included individuals ringed in Belgium (2), Netherlands, Denmark (2) and Northern Ireland (3) as well as 75 ringed in the UK (Ward 2000). All but one of these ringed birds from the continent were juveniles, suggesting that young birds are most likely to cross the North Sea during autumn dispersal. There is also one recovery of a juvenile reared at a colony in North America (so of a different subspecies from the birds in Europe) recovered dead in SW England in November (Wernham et al. 2002). The North American subspecies normally winters in South America, so this ring recovery is highly atypical. Many juveniles remain dependent on their parents for some of their food during migration and during winter (Fernandez-Cordeiro and Costas 1991; Wernham et al. 2002). Most first year birds remain in the winter quarters through their first summer and second years mainly move only part way towards their natal area, summering off west Africa or southern Europe. Most three year olds and older birds migrate rapidly back to their breeding area in March-April, but some three year olds, and some older birds spend the summer in west Africa or southern Europe rather than breeding (Wernham et al. 2002). Birds may recruit into colonies hundreds of kilometres from where they were reared, so there is considerable interchange between colonies in UK, Ireland and countries on the east side of the North Sea (Wernham et al. 2002). Seabird 2000 estimated that about 12,490 pairs bred in the UK, 1,800 pairs in Ireland, 4,500 in Denmark, 9,700 in Germany, 14,500 in The Netherlands, and 1,550 in Belgium (Mitchell et al. 2004). With extensive dispersal between these populations and the large numbers on each side of the North Sea, it is likely that many of the Sandwich terns in UK waters in July-October originate from mainland European colonies (and some also from Ireland though numbers there are relatively small). Few breed in Norway or Sweden (in total about 300 to 400 pairs) and there are none in Faroe or Iceland, so numbers migrating through UK waters from further north will be very small. Meissner and Krupa (2007) reported that Sandwich terns caught in the southern Baltic during migration had longer wing lengths than birds caught in NE England on migration, indicating that different populations were involved in these two regions. It is likely that the numbers of birds crossing the North Sea during post-breeding dispersal will vary considerably from year to year, as terns will congregate, post-breeding, in areas where there are aggregations of prey fish; small pelagic fish such as sandeels, sprats and young herring (Stienen and Brenninkmeijer 1998, 2002). Productivity of these short-lived fish varies considerably from year to year, and so there may be some years when many UK birds move to Danish waters to feed on sprats, some years when many Dutch birds move to UK waters to feed on sandeels, and so on. As a result, the proportions of birds from different countries and the absolute numbers of birds in UK waters post-breeding and during migration may vary considerably from year to year. Although large numbers of Sandwich terns breed in France (about 7,000 pairs) and many birds are ringed in those colonies, they are not recovered in the UK and so appear not to pass through UK waters. The distribution of Sandwich tern colonies in France is predominantly in the Bay of Biscay, with few nesting in northern France (Hagemeijer and Blair 1997), so the lack of connectivity with the UK is understandable.

16.7 Numbers in UK waters

Forrester et al. (2007) suggest that about 500 to 1,500 birds are in Scottish waters during autumn migration, and about 100 to 1,000 birds during spring migration, and that there may be up to 5 birds in Scottish waters in winter. Numbers in English waters are uncertain, but likely involve all of the UK population (of about 12,500 pairs so 25,000 adults). Associated with that UK adult population will be about 15,700 immatures, but the youngest age class will predominantly remain in the winter quarters rather than return to UK waters, so perhaps about 8,000 to 9,000 of the immatures are likely to be in UK waters during the migration periods. In addition, even more uncertain numbers from overseas populations pass through

UK waters on passage. These are likely to include about 1,000 to 4,000 birds from Ireland passing mainly through SW English waters, and perhaps 1,000 to 20,000 birds from Norway to Belgium passing mainly through southern North Sea UK waters (as many of those birds will pass through southern North Sea continental rather than UK waters). Summing these suggests that about 44,000 birds may pass through UK waters during autumn migration, and perhaps similar or slightly smaller numbers in spring.

16.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the European population, comprising 132,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 69,000-79,000 pairs. Kober et al. (2010) did not present an estimated biogeographic population for this species. The biogeographic population with connectivity to UK waters would be the sum of the populations listed in Figure 16.2, or a total of about 45,000 pairs. Populations in France (which are predominantly in the Bay of Biscay and western Mediterranean; Hagemeijer and Blair 1997) and Spain appear to have no connectivity with UK waters. The biogeographic population with connectivity to UK waters comprises 148,000 birds (adults and immatures) with 41,000 from UK and 107,000 from overseas. However, only a small proportion of the birds from the connected overseas populations occur within UK waters, so that the estimated total number of birds in UK waters during migration is 49,000 birds, with 35,300 from UK and 13,600 from overseas.



Figure 16.2. Breeding population origins of Sandwich terns in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

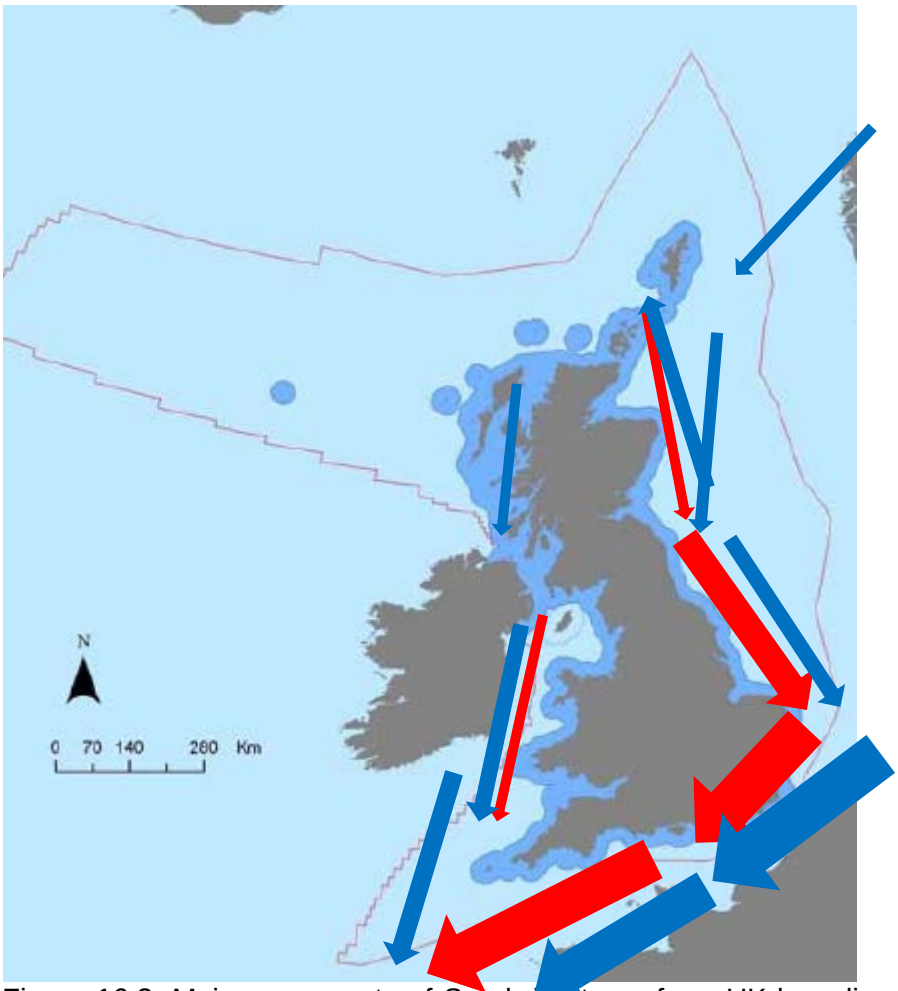


Figure 16.3. Main movements of Sandwich terns from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

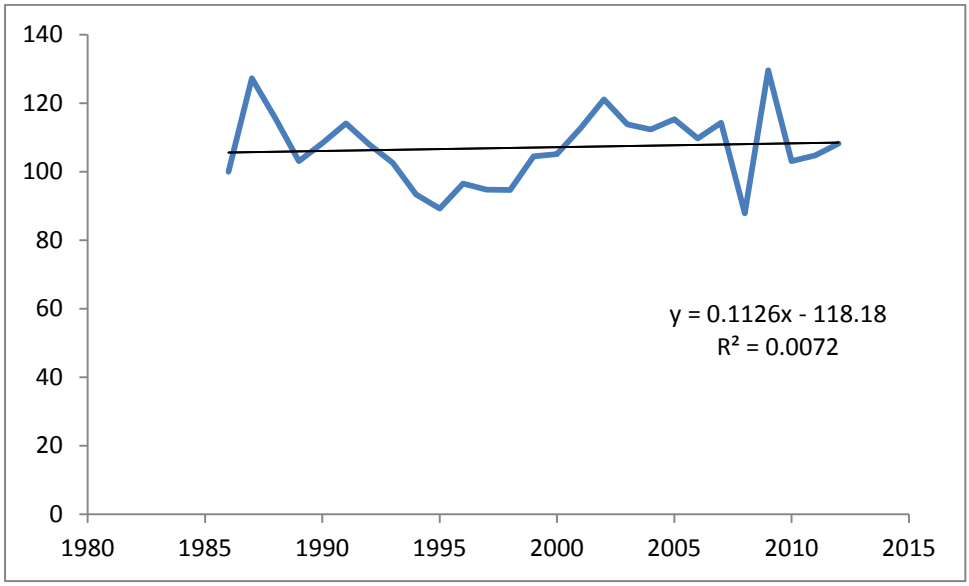


Figure 16.4. Trend in the Sandwich tern breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

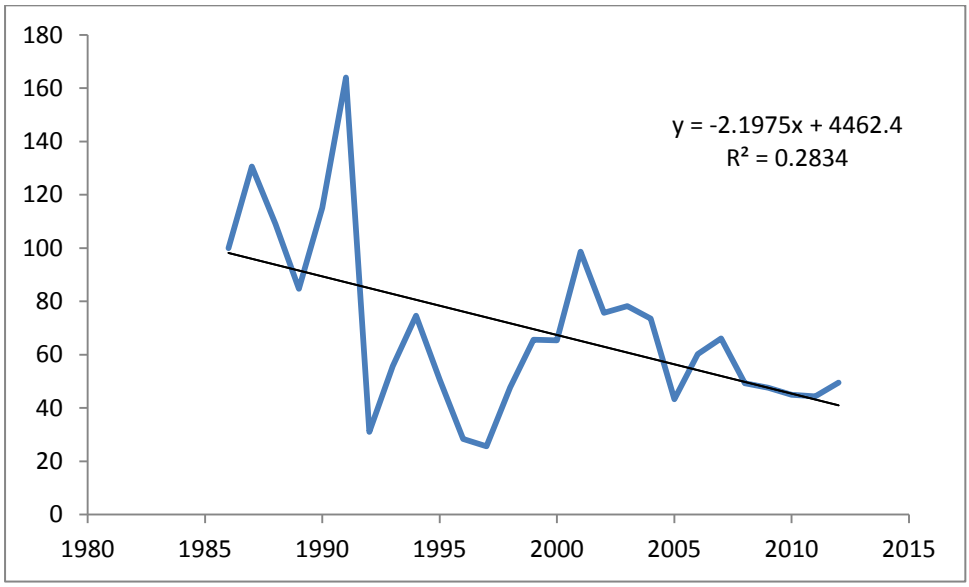


Figure 16.5. Trend in the Sandwich tern breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

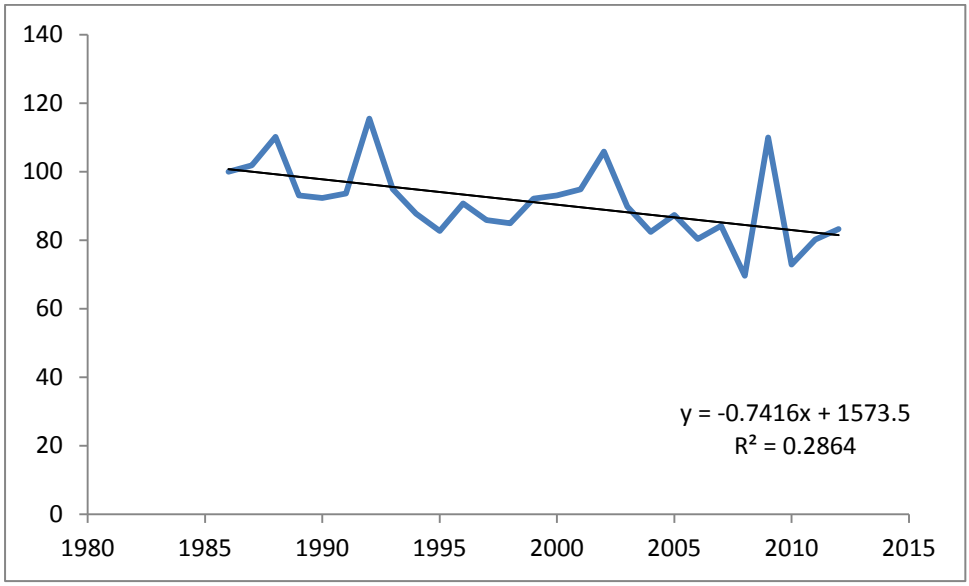


Figure 16.6. Trend in the Sandwich tern breeding population index in England from 1986-2012. Data from JNCC seabird population monitoring database.

16.9 Proportion of BDMPS from UK breeding SPAs

The 16 SPAs with breeding Sandwich terns as a feature together held 11,440 pairs at designation, estimated to represent ca. 72% of the British breeding population (Stroud et al. 2001). Based on census data from 2006-2011, Stroud et al. (2014) estimated that the population on GB SPAs for breeding Sandwich terns comprised 72%, suggesting no change overall in this statistic since SPA designations. This is despite the fact that several SPA populations have declined to zero (Table 16.1).

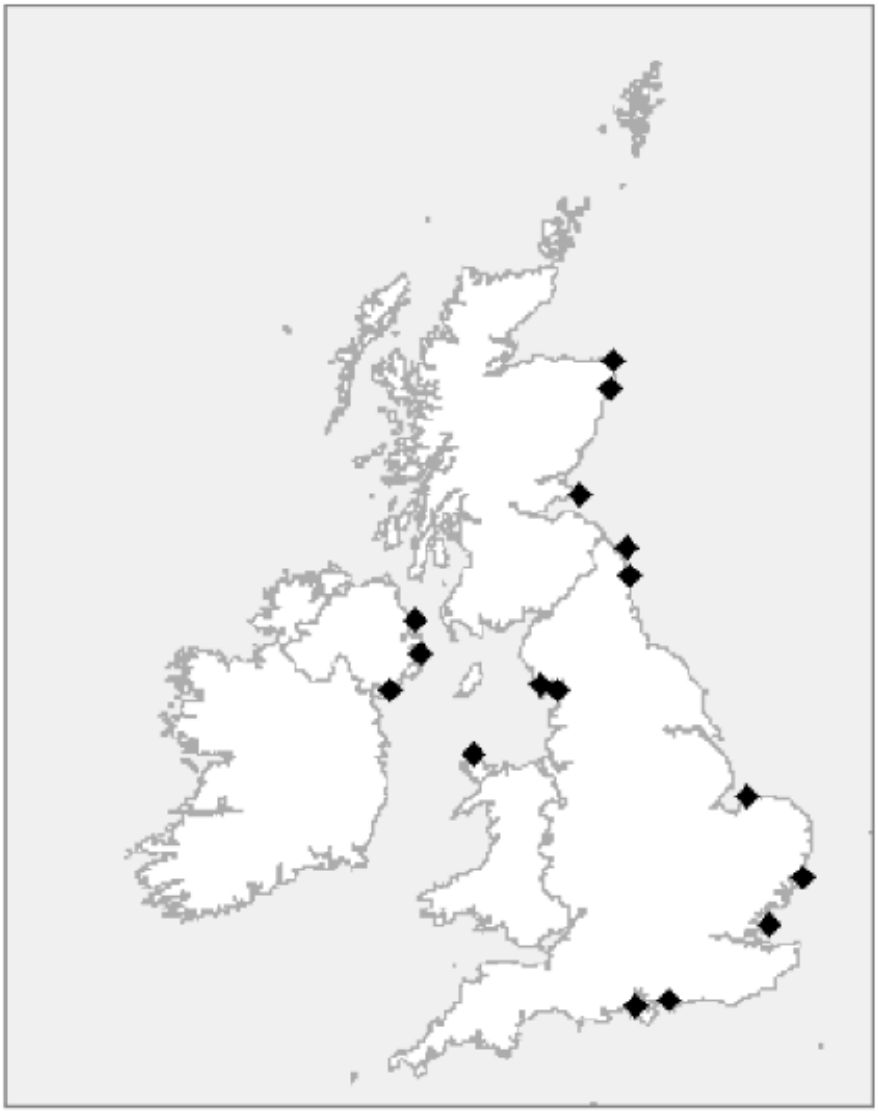


Figure 16.7. The UK SPA suite for breeding Sandwich terns. These SPA populations are listed in Table 16.1.

Table 16.1. The UK SPA suite for breeding Sandwich terns.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
Loch of Strathbeg	NE Scotland	530	1995	Declined 2004	0 0 0 0 0 0 0	2007 2008 2009 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database SMP database SMP database SMP database
Ythan Estuary, Sands of Forvie	NE Scotland	600 (early 1990s)	1998	Maintained 2012	900 670 645 674 590 657 565	2007 2008 2009 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database SMP database SMP database

Forth Islands	E Scotland	440 (1985) Or 22 (Stroud et al. 2001)	1990	Declined 2003	0 1 0 0 0 0	2007 2008 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database SMP database SMP database
Farne Islands	NE England	2,070 (1993-1997)	1985		1,413 1,358 1,415 1,019 544 966 824	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Coquet Island	NE England	1,590 (1993-1997)	1985		759 1,223 804 873 1,069 1,717 1,289 670	2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
North Norfolk Coast	E England	3,700 (1992-1996) Or 3,457 (Stroud et al. 2001)	1989		3,550 3,450 3,600 2,680 3,100 2,980 3,562 4,135	2005 2006 2007 2008 2009 2010 2011 2012	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Alde-Ore Estuary	E England	170 (1992-1996) Or 169 (Stroud et al. 2001)	1996		2 3 0 0 0 2	2004 2005 2006 2007 2008 2009	SMP database SMP database SMP database SMP database SMP database SMP database
Foulness		320 (1992-1996)	1996		0 0 0 0	2003 2004 2005 2006	SMP database SMP database SMP database SMP database
Chichester & Langstone Harb	S England	31 (1993-1997) Or 158 (Stroud et al. 2001)	1987		271 204 78 130 183 205 175 46 6	2005 2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Solent & Southampton Water	S England	231 (1993-1997)	1998		275 268 210 226 0 140 0 0 0 215 0	1999 2000 2001 2002 2003 2004 2005 2006 2007 2007 2008	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database Stroud et al. 2014 SMP database

UK Western waters							
Carlingford Lough	N Ireland	575 (1993-1997)	1998		1,125 826 363 170 0 78 0	2005 2006 2007 2008 2009 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Larne Lough	N Ireland	165 Stroud et al. 2001)	1997		788 465 695 545 373 449 324 433 257	2005 2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Strangford Lough	N Ireland	593 (1993-1997)	1998		1,092 1,385 1,594 1,398 1,994 1,203 978 771	2005 2006 2007 2008 2009 2010 2011 2012	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Morecambe Bay	NW England	422 (1992-1996) Or 290 (Stroud et al. 2001)	1996		0 0 0 0 0 1	2002 2003 2004 2005 2006 2011	SMP database SMP database SMP database SMP database SMP database SMP database
Duddon Estuary	Cumbria	210 (1988-1992)	1998		300 300 280 400 400 10 0 1	2005 2006 2007 2008 2009 2010 2011 2012	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Ynys Feurig, Cemlyn Bay	Wales	460 (1993-1997)	1992		0 0 1 0 0 0	2004 2005 2006 2007 2008 2009	SMP database SMP database SMP database SMP database SMP database SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

16.10 BDMPS

The UK waters can be divided into two distinct spatial BDMPS for Sandwich tern, the UK North Sea and Channel, and the UK western waters. These areas are appropriate for passage periods, including both autumn and spring. The UK North Sea and Channel BDMPS holds the bulk of the overseas migrants passing through UK waters and the bulk of the UK breeding population. About 38,000 birds may occur in this BDMPS in autumn and spring, with about 28,000 of those being from the UK population. The UK western waters

BDMPS holds smaller numbers of birds, with about 11,000 in total and 7,300 of these from the UK and 3,500 from overseas.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 51 and 52.

Based on evidence reviewed in sections 16.5, 16.6 and 16.7, the UK North Sea and Channel migration seasons BDMPS holds 100% of adults and 70% of immatures from UK North Sea colonies, none from UK western waters colonies, and 10% of adults and immatures from Norway, Sweden, Denmark, Germany, The Netherlands, and Belgium, but none from Ireland. This gives a BDMPS total of 38,051 birds, 27,961 from UK and 10,090 from overseas.

Based on evidence reviewed in sections 16.5, 16.6 and 16.7, the UK western waters migration seasons BDMPS holds 0% of adults and immatures from UK North Sea colonies, 100% of adults and 70% of immatures from UK western waters colonies, 5% of adults and immatures from Norway and Sweden, 3% of adults and immatures from Denmark, 2% of adults and immatures from Germany, 1% of adults and immatures from The Netherlands and Belgium, 30% of adults and immatures from Ireland. This gives a BDMPS total of 10,761 birds, 7,291 from UK and 3,470 from overseas.

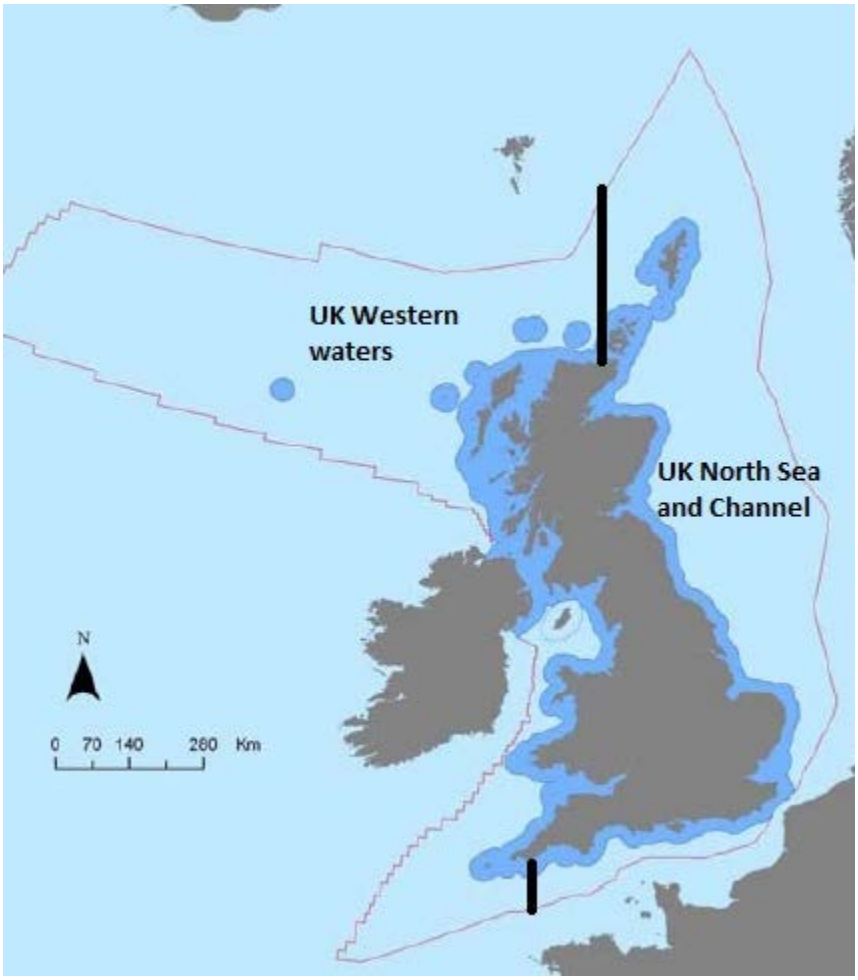


Figure 16.8. Two defined BDMPS spatial areas for Sandwich tern: 'UK North Sea waters and Channel' and 'UK Western waters'.

16.11 Proportions of UK SPA birds in BDMPS

The proportion of birds in each BDMPS that are adults from UK SPA populations can be estimated directly from Appendix A Tables 51 and 52. For example, the UK North Sea and Channel migration season BDMPS comprises 38,051 birds in total, of which 12,404 are adults from UK SPA populations, giving an estimate of 33% being adults from UK SPAs.

16.12 Spatial distribution of UK breeding SPA birds across the BDMPS

During migration periods, UK SPA birds will be fairly well mixed throughout the BDMPS area. In UK western waters the very high concentration of most SPA birds in a single SPA may result in some local aggregation of SPA birds around North Wales. However, dispersal of birds in autumn can be quite rapid so that aggregations of UK SPA birds are likely to disappear as migration proceeds.

17. ROSEATE TERN *Sterna dougallii*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (August-September and late April-May) (adults and immatures)
Overseas	2,600	2,111
UK	300	244
Total	2,900	2,355

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Migration seasons BDMPS (August-September and late April-May)			
East coast and Channel	251	7	244
North and west Scotland	4	4	0
West England & Wales	2,100	2,100	0

Although scarce, roseate tern is intensively monitored in the UK and Ireland. Colony locations are regularly checked, and breeding numbers are counted annually at most colonies. Migrations of roseate terns through UK waters have not been studied in detail, but it is certain that birds from UK colonies pass through UK waters on migration (apart from very young immatures that remain in the winter quarters throughout their first summer). It is almost certain that Irish roseate terns migrate through western UK waters, since they would have difficulty getting from Ireland to west Africa without passing through the SW Approaches. There is unlikely to be significant interchange between birds from western waters and the North Sea, as roseate terns are not seen migrating overland in the way that common terns often do. The main uncertainty is what proportion of immature roseate terns from the Irish population migrate through UK waters, and for that reason the numbers of overseas roseate terns in the West England & Wales BDMPS are coded amber, while other component numbers are coded green.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 53 to 55.

17.1 Breeding range and taxa

Roseate tern is a cosmopolitan species, breeding in tropical, sub-tropical and temperate regions around the world. There are five subspecies, but only nominate *dougallii* occurs in

British waters. The species is so scarce in the UK that useful biometrics are unlikely to be available.

17.2 Non-breeding component of the population

Roseate terns start to breed when 2 years old (BTO Birdfacts). Adult survival rate is 0.855 (BTO Birdfacts), juvenile survival unknown (BTO Birdfacts) and mean productivity is 1.293 chicks per pair (JNCC database, n=88 measurements). To obtain a stable population, productivity was adjusted to 1 chick per pair as the reported productivity seems out of line with other data on productivity of terms and may be biased by coming predominantly from highly protected colonies, survival of immatures was adjusted to 0.5 for juveniles, and 0.6 for 1-year olds. The model population comprised 57% adults, 29% juveniles and 14% older immatures. There are 0.75 immatures per adult.

17.3 Phenology

Breeding colonies in the UK are deserted by late August, with modal departure in August (Pennington et al. 2004; Forrester et al. 2007). Autumn migration starts in July (Wernham et al. 2002; Forrester et al. 2007) or late-August (Cramp et al. 1977-94). Peak autumn migration occurs in August (Forrester et al. 2007), August-September (Wernham et al. 2002), or September (Cramp et al. 1977-94). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in August, with very few after early September (Figure 17.1). Autumn migration is completed by early October (Forrester et al. 2007) mid-October (Cramp et al. 1977-94) or October (Wernham et al. 2002).

Spring migration starts in late March from southern hemisphere wintering areas (Cramp et al. 1977-94), late April (Forrester et al. 2007) or early May (Wernham et al. 2002) in UK waters. Peak spring migration occurs in May (Cramp et al. 1977-94; Wernham et al. 2002; Brown and Grice 2005; Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in mid- to late-May (Figure 17.1). Spring migration is completed by early June (Cramp et al. 1977-94; Wernham et al. 2002) or June (Forrester et al. 2007). Birds re-occupy colonies from early May, with modal return in mid- to late-May (Forrester et al. 2007). It is interesting to note that numbers seen on spring migration are very much smaller than numbers seen on autumn migration (Figure 17.1). This pattern is typical of most seabird species but is very pronounced for roseate tern. The reasons for this are not understood. The fact that spring migration occurs more rapidly than autumn migration may be a major factor. Possibly the fact that autumn migration includes juvenile birds may also be a factor (since the inexperienced juveniles may be particularly evident passing coastal migration watch points in autumn as they might perhaps migrate closer to shore than most adults do).

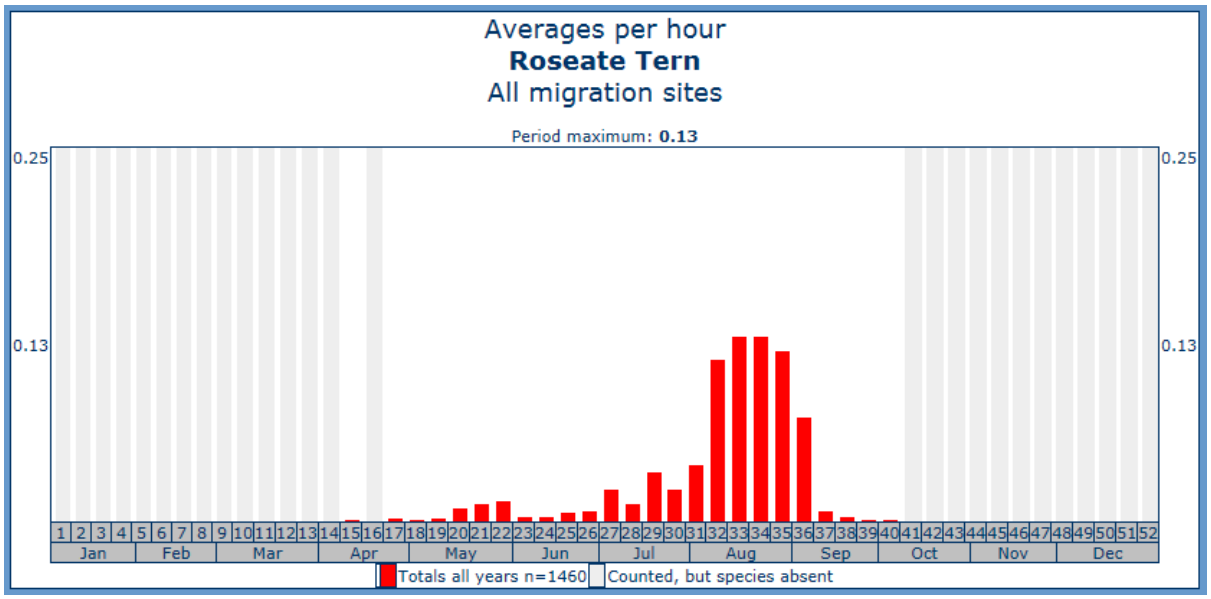


Figure 17.1. Average numbers of roseate terns counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season birds predominantly absent from UK waters. From the data reviewed above, an appropriate definition would be breeding season May-August, non-breeding season September-April.

17.4 Defined seasons:

- UK Breeding season — May-August
- Post-breeding migration in UK waters — August-September (**migration BDMPS**)
- non-breeding season — September-April
- Return migration through UK waters late — April-May (**migration BDMPS**)
- Migration-free breeding season — June-July
- Migration-free winter season — October-March

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for roseate tern:

Migration periods BDMPS (August-September, and late April-May).

17.5 Movements of birds from the UK population

Roseate terns at UK colonies fledge chicks in July, and pre-migratory dispersal occurs in August (Wernham et al. 2002). At this time, birds tend to congregate where there is suitable food, and chicks remain dependent on their parents for feeding (Wernham et al. 2002). Autumn migration to wintering areas off west Africa occurs mainly during August-October, although some birds (presumably failed breeders or nonbreeders) arrive on the wintering grounds by July (Wernham et al. 2002). Almost all juveniles remain on the wintering grounds through their first summer, although very small numbers return to visit breeding colonies briefly in July. Many, but not all, 2nd year birds return to breeding areas in late June and July to prospect for nest sites. Older birds leave west Africa in March-April and return to colonies in May (Wernham et al. 2002).

17.6 Movements of birds from overseas into UK waters

No roseate terns ringed at colonies outside the British Isles have been recovered within UK waters (Wernham et al. 2002). However, although there is a preference for returning to the natal colony, chicks are likely to recruit into any colony in NW Europe, so this population clearly represents a meta-population with extensive gene flow. In contrast, chicks from NW Europe have hardly ever been seen in colonies in the Azores (where there are between 1,000 and 1,500 pairs) or North America (where there are around 4,000 pairs), suggesting that those populations are somewhat distinct. Seabird 2000 recorded about 56 pairs breeding in the UK, 734 in Ireland, 80 in France, and 1-3 pairs in Germany, Netherlands and Belgium (Mitchell et al. 2004). Based on the much larger numbers breeding in Ireland than in the UK, it seems likely that a very high proportion of the roseate terns seen in UK waters to the west of the UK in spring or autumn will be Irish birds (Brown and Grice 2005). Most UK roseate terns breed on the coast of Northumberland (colonies in the Firth of Forth which used to be a stronghold have declined to just one or two pairs since 2000). A high proportion of roseate terns in North Sea UK waters are likely to be from UK colonies as there is no evidence to suggest that Irish (or French) roseate terns pass through the North Sea.

17.7 Numbers in UK waters

Numbers in UK waters are very low, and so are very difficult to assess with any confidence. Forrester et al. (2007) suggest that between 5 and 20 birds migrate through Scottish waters. The relatively large population breeding in Ireland (750 pairs plus some of the associated immatures) almost certainly passes though SW English waters during autumn and spring migrations.

17.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the European population, comprising 1,770 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 1,900-2,400 pairs. Kober et al. (2010) did not present an estimated biogeographic population for this species. The biogeographic population with possible connectivity to UK waters comprises 84 pairs in the UK, 750 pairs in Ireland, and 3 pairs in Germany to Belgium (Figure 17.2). This equates to 2,900 birds in total, with 300 from UK and 2,600 from overseas. A high proportion of this biogeographic population with connectivity does pass through UK waters on migration. Estimated numbers in UK waters during migration are 2,340 birds in total, with 240 from UK and 2,100 from overseas (the total from UK in UK waters is less than the biogeographic total in the UK population because some first year birds remain in winter quarters so do not enter UK waters at that stage of their life).



Figure 17.2. Breeding population origins of roseate terns in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

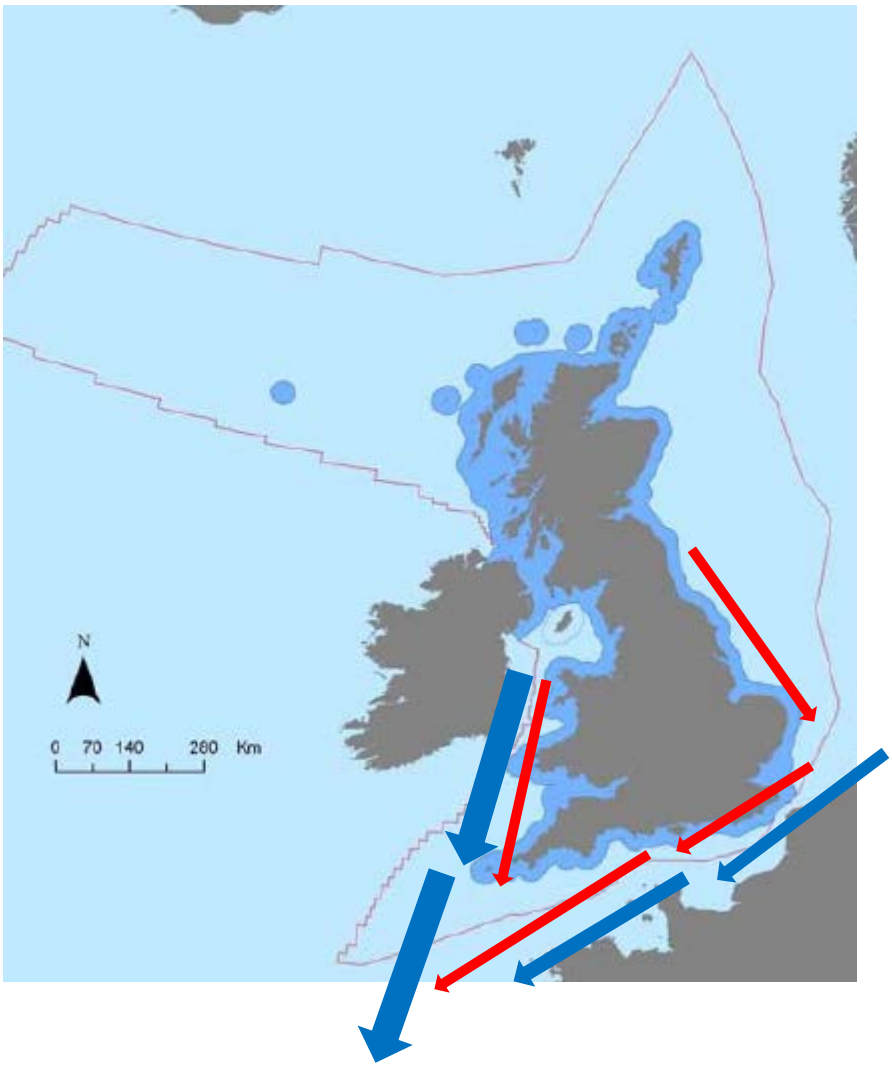


Figure 17.3. Main movements of roseate terns from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

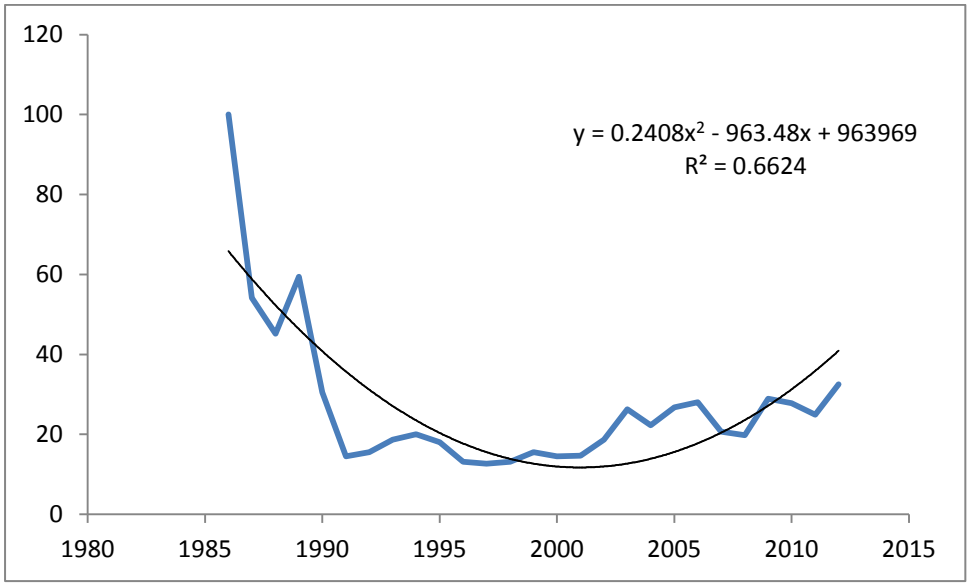


Figure 17.4. Trend in the roseate tern breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

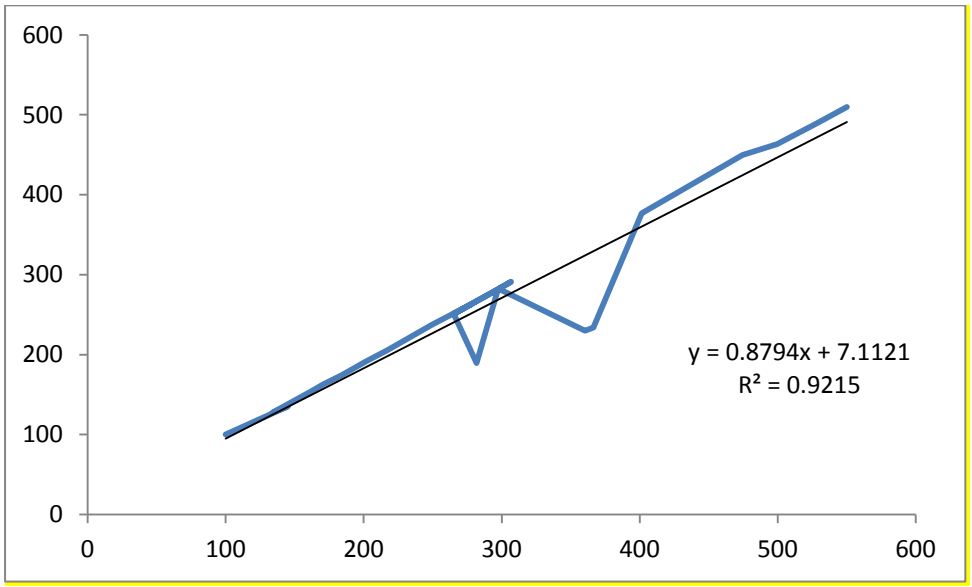


Figure 17.5. Trend in the roseate tern breeding population index in all-Ireland from 1986-2012. Data from JNCC seabird population monitoring database.

17.9 Proportion of UK population from UK breeding SPAs

The 7 SPAs with breeding roseate terns as a feature together held 56 pairs at designation, estimated to represent ca. 88% of the British breeding population and 1.4% of the all-Ireland breeding population (Stroud et al. 2001). Stroud et al. (2014) estimate that the UK SPA populations counted in 2005-2011 represented 94% of the GB population.



Figure 17.6. SPA suite for roseate tern. These SPA populations are listed in Table 17.1.

Table 17.1. The UK SPA suite for breeding roseate terns.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
East coast and Channel							
Forth Islands	E Scotland	8 (1997-2001) Or 9 (Stroud et al. 2001)	1990 (and 2004)	Declined 2009	3	2005-2009	Stroud et al. 2014
Farne Islands	NE England	3 (Stroud et al. 2001)	1985		0	2011	Stroud et al. 2014
Coquet Island	NE England	31 (1993-1997)	1985		78	2011	Stroud et al. 2014
North Norfolk Coast	E England	2 (Stroud et al. 2001)	1989		0	2010	Stroud et al. 2014

Solent & Southampton Water	S England	2 (1993-1997)	1998		0	2009	Stroud et al. 2014
West England & Wales							
Larne Lough	N Ireland	6 (1993-1997)	1997		0	2011	Stroud et al. 2014
Ynys Feurig, Cemlyn Bay	Wales	3 (1992-1996)	1992		0	2011	SCM database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

17.10 BDMPS

UK waters can be divided into three BDMPS based on strong differences in origins and numbers of birds present in the three areas during migration seasons. 'North and West Scotland' holds no breeding birds and has a BDMPS of about 4 birds, which are most likely to be immatures from the Irish population. 'East Coast and Channel' holds a breeding population of about 82-84 pairs, of which 81 are in SPAs. The BDMPS comprises these 82-84 pairs plus associated immatures, plus about 7 birds from the population in Germany to Belgium that may pass through UK waters. In total this BDMPS probably includes 251 birds. The 'West England and Wales' BDMPS holds no UK breeding birds, but will see migration of many birds from the population in Ireland. Possibly some 2,100 roseate terns migrate to and from the east coast of Ireland through the West of England and Wales marine area. So the BDMPS for this area is 2,100 birds, all from outwith the UK population.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 53 to 55.

Based on evidence reviewed in sections 17.5, 17.6 and 17.7, the East coast and Channel migration seasons BDMPS holds 100% of adults and 60% of immatures from UK North Sea colonies but no birds from other parts of the UK, 5% of adults and 10% of immatures from Germany, The Netherlands and Belgium, and 0.2% of adults and 0.3% of immatures from Ireland (Appendix A Table 53). These proportions result in a BDMPS population total of 251 birds, 244 from UK and 7 from overseas.

Based on evidence reviewed in sections 17.5, 17.6 and 17.7, the North and West Scottish waters migration seasons BDMPS holds no birds from UK colonies, but 0.1% of immatures from Germany, The Netherlands and Belgium, and 0.05% of adults and 0.3% of immatures from Ireland (Appendix A Table 54). This gives an estimated BDMPS of 4 birds, all from overseas populations.

Based on evidence reviewed in sections 17.5, 17.6 and 17.7, the West England and Wales migration seasons BDMPS holds no birds from UK North Sea colonies, 100% of adults and 60% of immatures from UK west coast colonies, 0.01% of immatures from Germany, The Netherlands and Belgium, 95% of adults and 60% of immatures from Ireland (Appendix A Table 55). This gives an estimated BDMPS of 2,100 birds, none from UK colonies but 2,100 from overseas colonies.

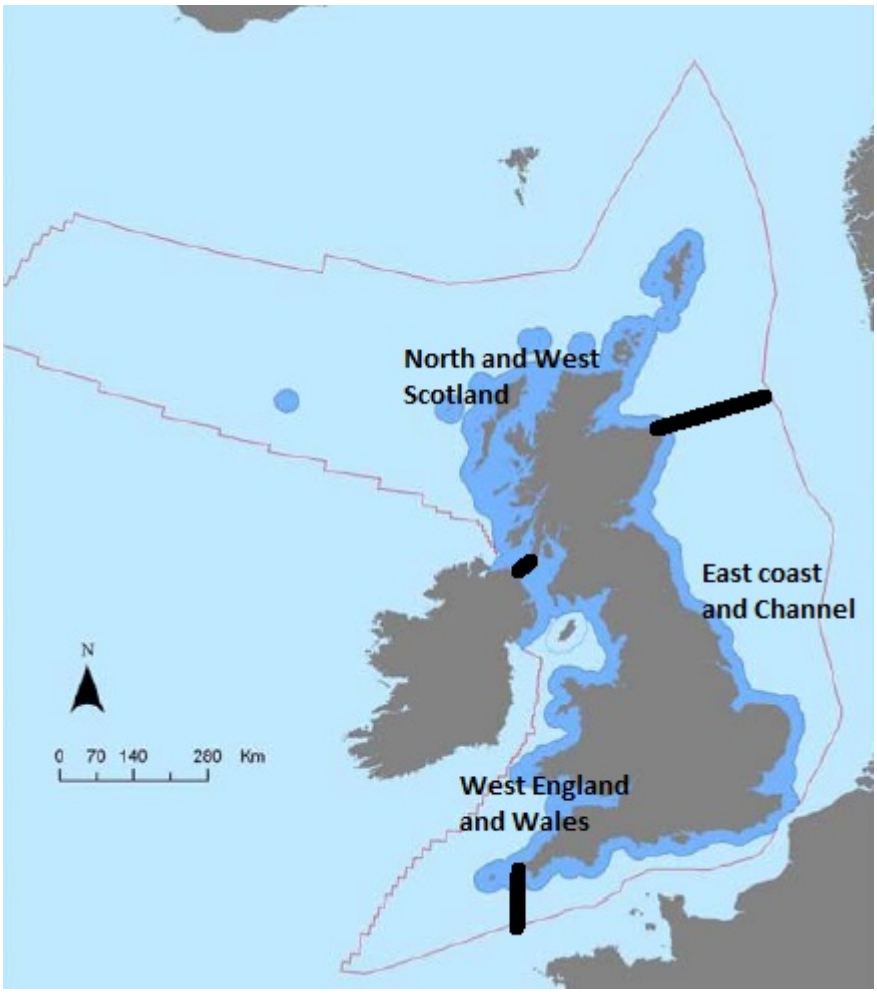


Figure 17.7. Three defined BDMPS spatial areas for roseate tern: ‘East coast and Channel’, ‘North and West Scotland’ and ‘West England and Wales’.

17.11 Proportions of UK SPA birds in BDMPS

In North and West Scotland BDMPS there are probably no UK SPA birds. In East Coast and Channel BDMPS UK birds are likely to form 97% of the population, with 94% of those 97% being UK SPA birds, so that UK SPA birds represent 91% of the population. In West England and Wales BDMPS UK birds are likely to form 0% of the population. The proportion of birds in each BDMPS that are adults from UK SPA populations can be estimated directly from Appendix A Tables 53 to 55. For example, the East coast and Channel migration season BDMPS comprises 251 birds in total, of which 168 are adults from UK SPA populations, giving an estimate of 67% being adults from UK SPAs.

17.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Since the UK SPA birds either form 0% or a high percentage of the BDMPS, the spatial distribution within regions is likely to be consistent; high in East coast and Channel BDMPS and zero in North and West Scotland BDMPS and in West England and Wales BDMPS.

18. COMMON TERN *Sterna hirundo*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (late July to early September, and April-May) (adults and immatures)
Overseas	440,000	174,416
UK	40,000	35,154
Total	480,000	209,570

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Migration seasons BDMPS (late July to early September and April-May)			
UK North Sea and Channel	144,911	125,969	18,942
UK Western waters	64,659	48,447	16,212

Common tern numbers in most UK SPA colonies are monitored frequently. However, numbers in UK colonies that are not SPA populations are less well monitored, and do represent a substantial proportion of the UK total. Common tern migrations have not been studied by geolocator deployment, and ringing recoveries from the migration period in UK waters are very limited. So understanding of details of common tern movements is relatively poor, especially to the extent that birds from overseas populations are concerned. While ring recoveries show that many birds from overseas pass through UK waters, the proportions of those populations doing so are very uncertain since ring recovery data are subject to considerable potential bias. Furthermore, these overseas populations are large, and certainly represent a high proportion of the total of common terns in UK waters during the migration season. There is yet another complication, which is that common terns rather frequently will migrate overland, and there is known to be considerable movement from North Sea estuaries over to western waters in autumn, and overland from southern England in spring. Therefore, estimated numbers of birds in the BDMPS are classed as red for the total population and numbers from overseas, whereas numbers from the UK population are classed amber.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 56 and 57.

18.1 Breeding range and taxa

Common tern has a Holarctic breeding range, predominantly in temperate latitudes. There are four subspecies, but only nominate *hirundo* occurs in British waters. Subspecies *hirundo*

breeds in North America, northern South America, the Atlantic Islands, most of Europe, north and west Africa, and through the Middle East to central Russia. Despite this large range, there appears to have been no assessment of whether biometrics would allow origins of individuals to be identified (Ward 2000). However, timing of primary moult varies between populations and can help to infer origins of birds caught on autumn migration (Ward 2000).

18.2 Non-breeding component of the population

Common terns start to breed when 3 years old (BTO Birdfacts). Adult survival rate is 0.9 (BTO Birdfacts), juvenile survival 0.47 up to 2 years old (BTO Birdfacts) and mean productivity is 0.721 chicks per pair (JNCC database, n=246 measurements). To obtain a stable population, survival of immatures was adjusted to 0.5 for juveniles, 0.7 for 1-year olds, and 0.8 for 2-year olds. The model population comprised 60% adults, 22% juveniles and 18% older immatures. There are 0.67 immatures per adult.

18.3 Phenology

Breeding colonies in the UK are deserted by late August, with modal departure in early August (Pennington et al. 2004; Forrester et al. 2007). Autumn dispersal/migration starts in early July (Pennington et al. 2004), mid-July (Cramp et al. 1977-94) or July (Wernham et al. 2002; Forrester et al. 2007). Peak autumn migration occurs in early August in Shetland (Pennington et al. 2004), in August (Forrester et al. 2007), August-September in UK waters in general (Wernham et al. 2002) and in Belgium (Vanermen et al. 2013). Peak migration through southern Europe and past west Africa continues through October (Cramp et al. 1977-94). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in August, with numbers declining rapidly in early September (Figure 18.1). Autumn migration is completed in Shetland by early September (Pennington et al. 2004), in UK waters by early October (Brown and Grice 2005; Forrester et al. 2007) or October (Wernham et al. 2002).

Spring migration starts in mid-March in the wintering areas of the southern hemisphere (Cramp et al. 1977-94), but starts in UK waters in early April (Wernham et al. 2002) or mid- to late-April in Shetland and Scotland (Pennington et al. 2004; Forrester et al. 2007). Peak spring migration occurs in early to mid-April in English waters (Brown and Grice 2005), April (Cramp et al. 1977-94), in April-May in Belgium (Vanermen et al. 2013), in April-May in UK waters (Wernham et al. 2002) and in early May in Shetland (Pennington et al. 2004). Forrester et al. (2007) suggests peak spring migration occurs in June, which seems rather late. Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late April and early May (Figure 18.1). Spring migration is completed by late May (Pennington et al. 2004), early June (Cramp et al. 1977-94), June (Wernham et al. 2002) or late June (Forrester et al. 2007). It is interesting to note that numbers seen on spring migration are very much smaller than numbers seen on autumn migration (Figure 18.1). This pattern is typical of most seabird species but is very pronounced for common tern. The reasons for this are not understood but seem to be due to behaviour of birds rather than to differences in numbers present. The fact that spring migration occurs more rapidly than autumn migration may be a major factor; if birds spend ten times longer on autumn migration through UK waters than on spring migration through UK waters it would be reasonable to expect counts at Trektellen sites to be ten times higher in autumn than in spring even if numbers of birds involved were the same. Possibly the fact that autumn migration includes juvenile birds may also be a factor (since the inexperienced juveniles may be particularly evident passing coastal migration watch points in autumn as they might perhaps migrate closer to shore than most adults do).

The first spring records of common tern in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 10 April to 20 May, but mostly in late April, and the last records were from 24 August to 30 October, but mostly in late September. Peak autumn

migration was reported in July-August in most years, and peak spring migration was reported in May (and usually in early May) in most years. Birds re-occupy colonies from late April, with modal return in mid- to late-May (Pennington et al. 2004; Forrester et al. 2007).

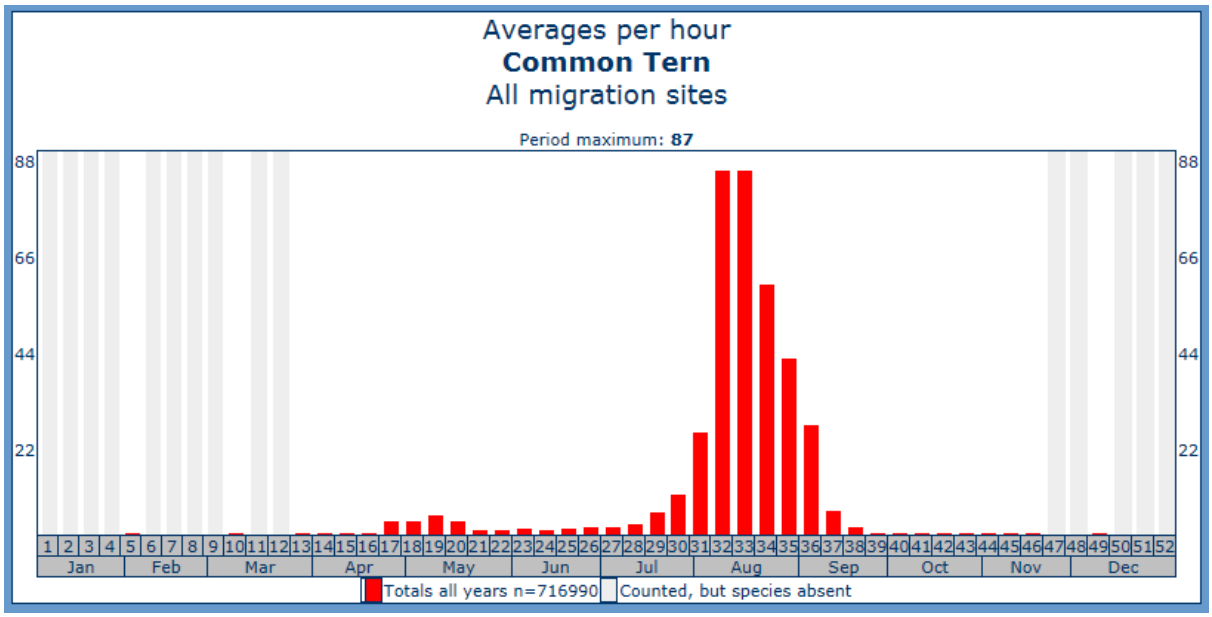


Figure 18.1. Average numbers of common terns counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-September, non-breeding season birds predominantly absent from UK waters. However, from the data reviewed above, a more appropriate definition would be breeding season May-August, non-breeding season September-April.

18.4 Defined seasons:

- UK Breeding season — May-August
- Post-breeding migration in UK waters — late July-early September (**migration BDMPS**)
- non-breeding season — September-April
- Return migration through UK waters — April-May (**migration BDMPS**)
- Migration-free breeding season — June-mid-July
- Migration-free winter season — October-March

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for common tern:

Migration periods BDMPS (late July-early September, and April-May).

18.5 Movements of birds from the UK population

Post-fledging dispersal from UK (and continental) colonies begins in July, but continues as late as October (Wernham et al. 2002). Post-fledging dispersal may be northwards rather than southwards, and may involve birds crossing the North Sea. For example, a fledgling ringed in Belgium was recovered in Durham together with fledglings from colonies in Norfolk in late August/early September (Wernham et al. 2002). As with many other tern species, fledglings tend to congregate in areas where feeding is easy (especially in estuaries and

large bays where there are presumably aggregations of sprats or sandeels), and may remain dependent on their parents for food for some time after fledging (Johnasson and Jakobsson 1997; Newton 2010), although they become independent of parents more quickly than Sandwich tern fledglings (Meissner and Krupa 2007). Some birds travel quickly to Africa, arriving in west Africa by August, while others remain in UK waters into September. During September and October, a strong southward migration occurs out of UK waters and along the coast of SW Europe to west Africa, with juveniles often still being fed by their parents. Migration follows the coastline (Wernham 2002). British birds appear to move south somewhat earlier than those from Norway, with those from Baltic colonies later still (Ward 2000).

18.6 Movements of birds from overseas into UK waters

Birds from many northern European countries pass through UK waters during post-fledging dispersal and autumn migration. Some birds move overland rather than following coasts, recognised routes being between the Firth of Forth and Clyde (Forrester et al. 2007) and between Teesmouth and Merseyside (Ward 2000). Ward (2000) reported peak numbers of common terns at Teesmouth in mid-August, with ringed fledglings from Lithuania (2), Finland (6), Sweden (2), Norway (9), and Netherlands (2) as well as 32 ringed as chicks at UK colonies. Based on moult scores, Ward (2000) inferred that a substantial minority of the adult common terns at Teesmouth in August were from the Baltic population, but the analysis was unable to estimate an accurate proportion because differences in timing of moult of UK and Baltic breeders are not well enough known. Wernham et al. (2002) report 101 ring recoveries to or from countries to the north and east, with 23 involving Belgium and the Netherlands, 14 involving Germany, Poland and the Baltic States, and 64 involving Fennoscandia. In contrast, there is no evidence from ringing of any movement of common terns from southern or eastern populations through UK waters. Common terns from North America are extremely rare visitors to Europe, and there are no records of American common terns reaching UK (Wernham et al. 2002).

Many adults return to breeding areas in the UK by April, and it is thought that spring migration is rapid and often occurs overland rather than tracking coasts (Wernham et al. 2002) At Dungeness and Portland Bill, spring passage of common terns peaks in late April and early May, and since movement is primarily eastwards at those sites and occurs at a time when many UK birds are already back at their colonies, probably involves birds returning to colonies in Fennoscandia or the Baltic States rather than to UK colonies (Wernham et al. 2002). However, migration timing may alter with climate change and oceanographic system oscillations (Favero et al. 2006). Although most first year birds remain in the wintering areas during the summer, most two year olds return to colonies, though they arrive from late May to late June. Three year olds often recruit into their natal colony, but substantial numbers may recruit elsewhere, with occasional movements to colonies in another country. In contrast, breeding adults are highly philopatric, usually returning to the same nest site in successive years, although there are a few cases of breeding dispersal to colonies across the North Sea (Wernham et al. 2002).

Based on data for Seabird 2000 (Mitchell et al. 2004), the UK breeding population (11,838 pairs) is small compared to some of the populations that may at least in part migrate through UK waters which total about 140,000 pairs (Finland 50,000, Sweden 22,000, Netherlands 19,000, Norway 15,000, Baltic States 12,750, Germany 9,000, Poland 6,000, Ireland 2,700, Belgium 2,250, Denmark 1,000). So it is likely that in August-October and in April-May, a substantial proportion of common terns in UK waters originate from these foreign populations. Meissner and Krupa (2007) reported that common terns caught in the southern Baltic during migration had longer wing lengths than birds from British breeding sites or birds caught in NE England on migration, indicating that different populations were involved in these two regions.

18.7 Numbers in UK waters

Numbers of migrating terns are difficult to assess. Forrester et al. (2007) suggest that between 2,000 and 20,000 are in Scottish waters during migration periods. However, migration through English waters will almost certainly include all of the UK breeding population (24,000 adults) plus some of the associated immatures (perhaps 8,000). It is also certain that large numbers of birds from continental Europe pass through UK waters, involving many tens of thousands of birds.

18.8 Biogeographic population and relevant smaller units (BDMPS)

Stroud et al. (2001) defined the biogeographic breeding population as that of the European population, comprising 195,105 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 220,000-340,000 pairs. Kober et al. (2010) did not present an estimated biogeographic population for this species. The biogeographic population with possible connectivity to UK waters (Figure 18) includes the 12,000 breeding pairs in the UK plus associated immatures (of the immature population of about 16,000 birds about half are likely to remain in the wintering area so will not pass through UK waters during migration periods). However, overseas populations with possible connectivity to UK waters sum to over 130,000 pairs plus associated immatures. This gives an estimated biogeographic population with connectivity to UK waters of 480,000 birds (adults and immatures), of which 40,000 are from the UK and 440,000 from overseas populations. So birds in UK waters during migration may include very large numbers from overseas. Unfortunately it is very uncertain how many of those overseas birds move through UK waters. The best available data suggest that there bare about 209,000 common terns (adults and immatures) in UK waters during migration, with 35,000 of these being from the UK population and 174,000 from overseas populations. Not all birds from the UK population are in UK waters during migration because many young immature birds remain in the winter quarters through their first summer.



Figure 18.2. Breeding population origins of common terns in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

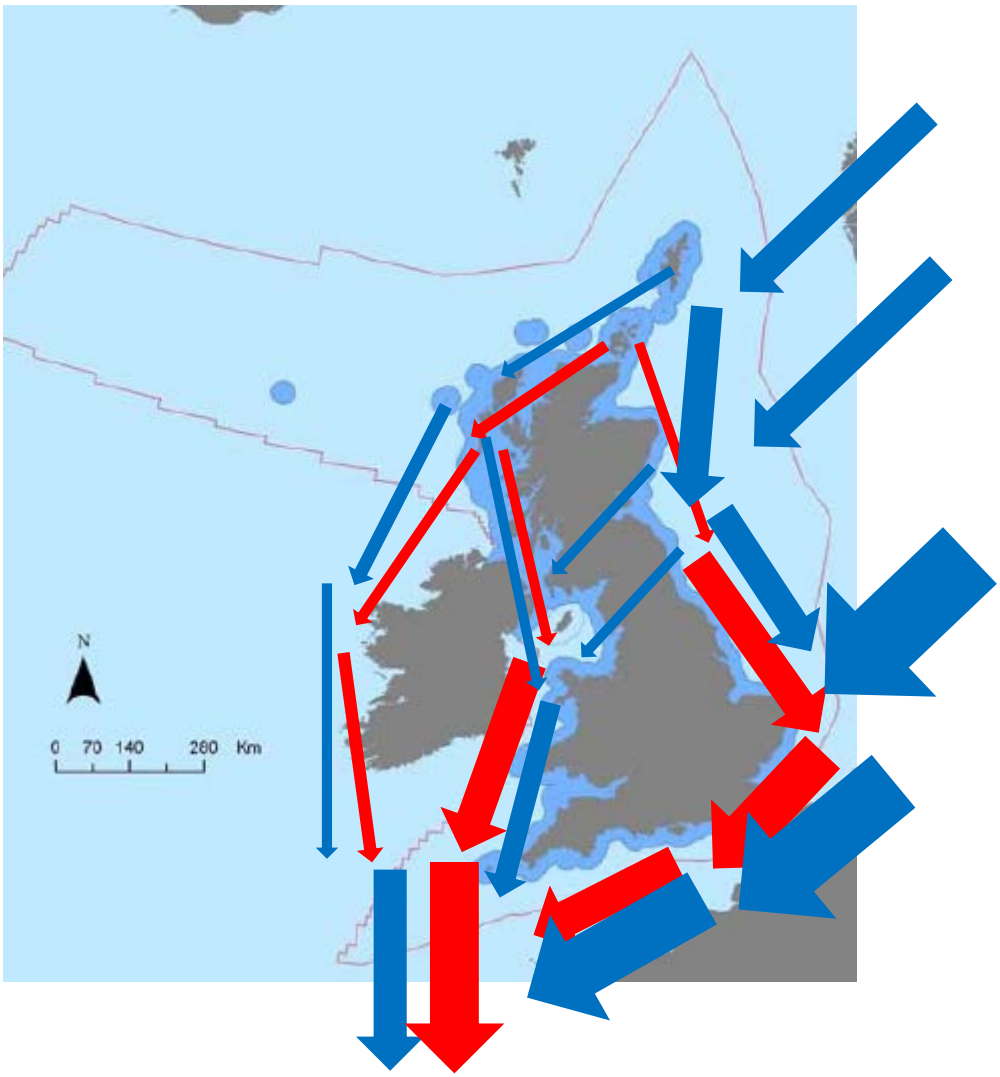


Figure 18.3. Main movements of common terns from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines, but for this species arrows that cross land do imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

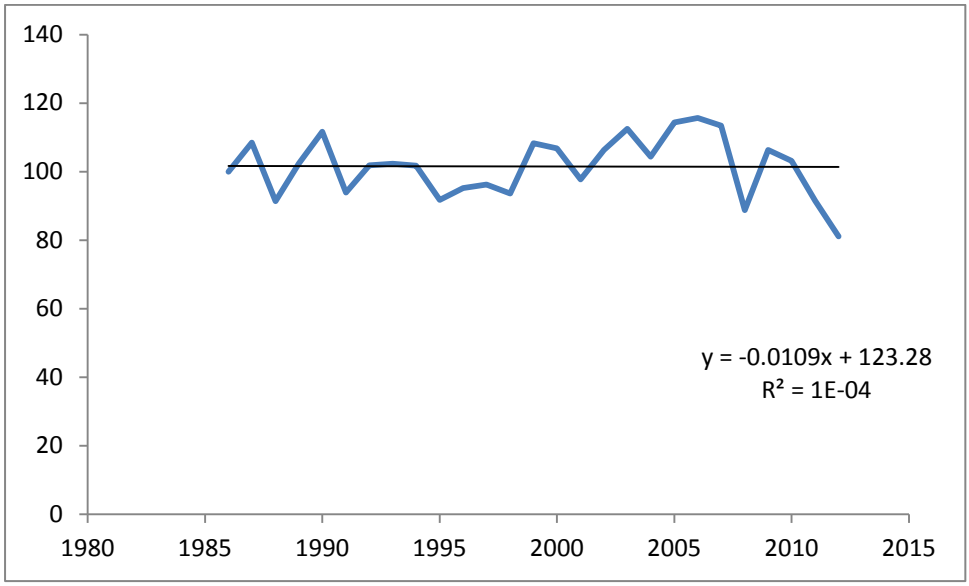


Figure 18.4. Trend in the common tern breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

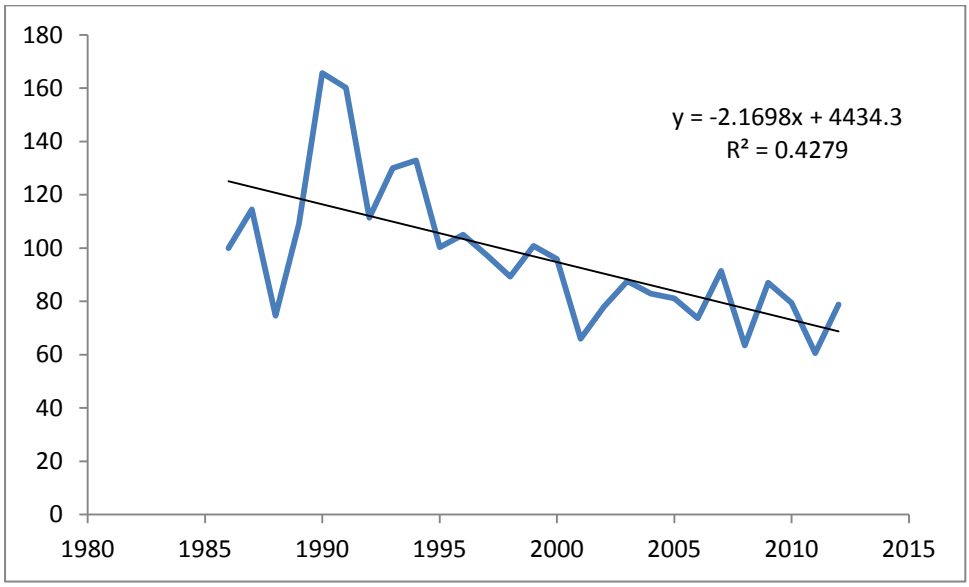


Figure 18.5. Trend in the common tern breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

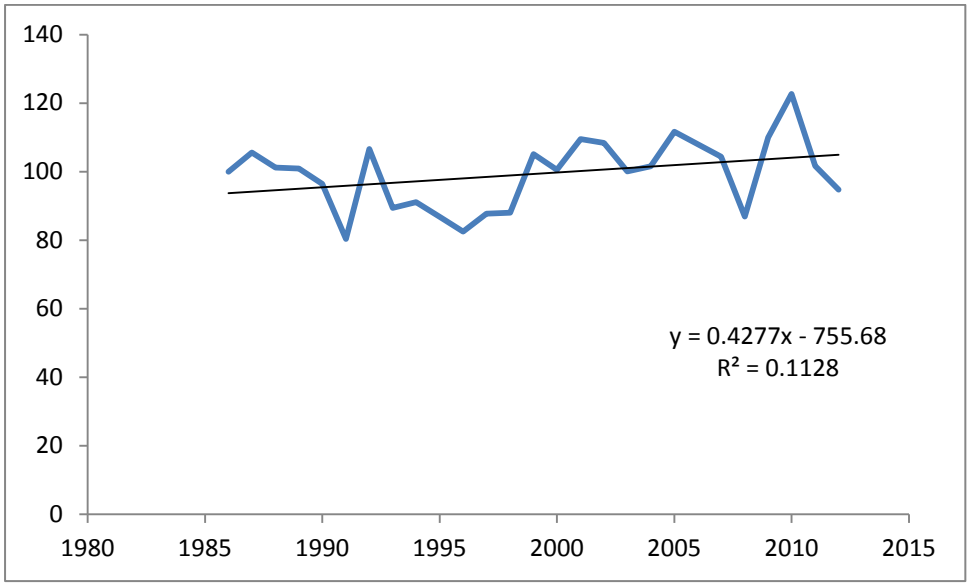


Figure 18.6. Trend in the common tern breeding population index in England from 1986-2012. Data from JNCC seabird population monitoring database.

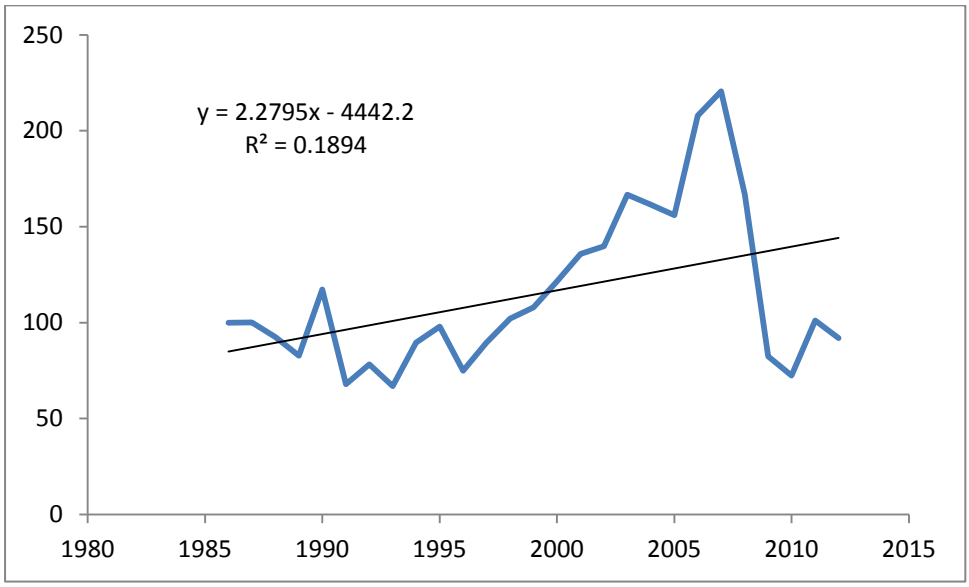


Figure 18.7. Trend in the common tern breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

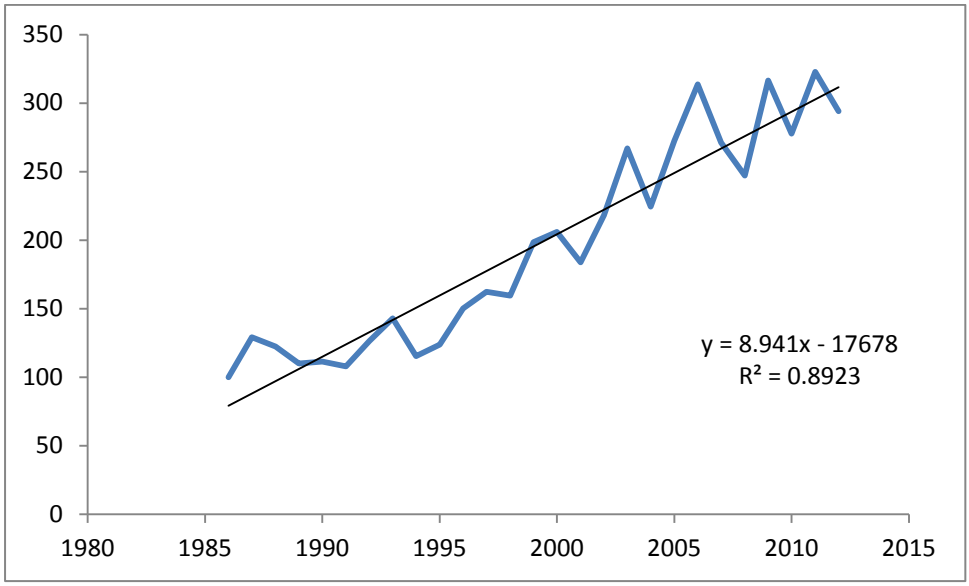


Figure 18.8. Trend in the common tern breeding population index in all-Ireland from 1986-2012. Data from JNCC seabird population monitoring database.

18.9 Proportion of UK population in UK breeding SPAs

The 23 SPAs with breeding common terns as a feature together held 7,551 pairs at designation, estimated to represent ca. 48% of the British breeding population and 42% of the all-Ireland breeding population (Stroud et al. 2001, updated to add Imperial Dock Lock SPA). Stroud et al. (2014) estimated that the GB SPA suite for breeding common terns held 43.8% of the GB population based on counts in 2007-2011.

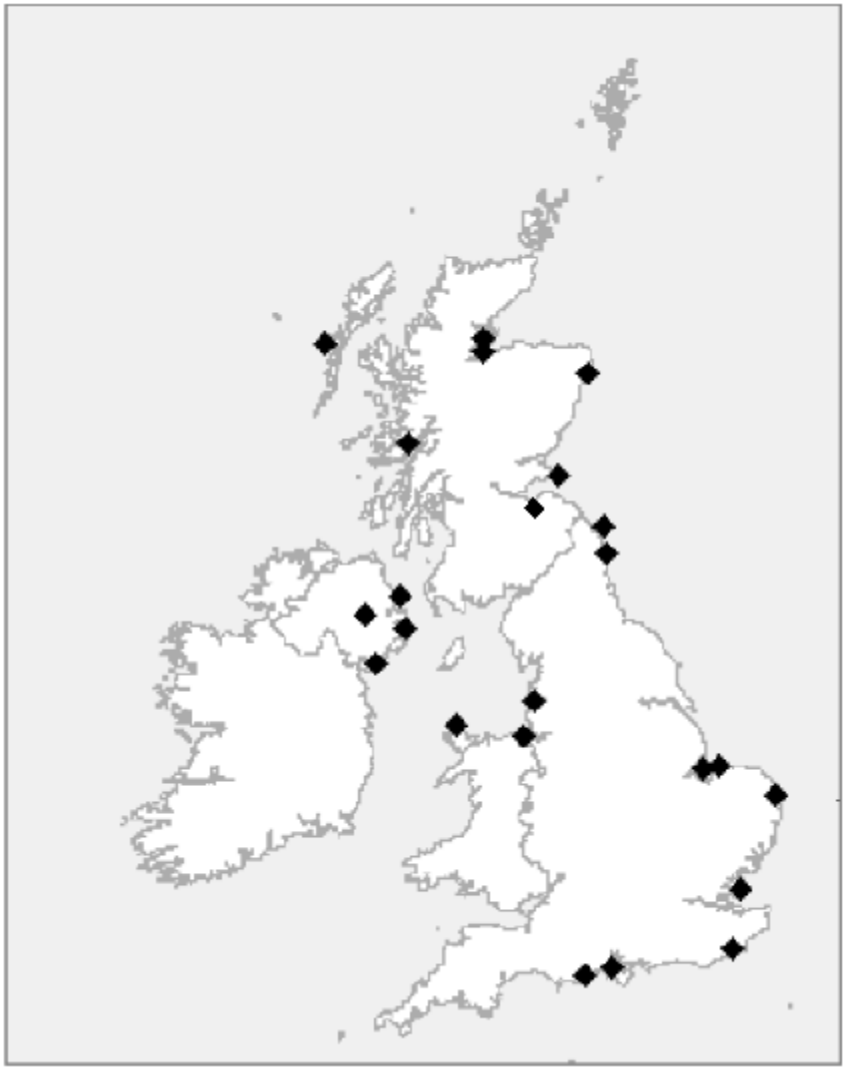


Figure 18.9. UK SPA suite for breeding common terns. These SPA populations are listed in Table 18.1.

Table 18.1. The UK SPA suite for breeding common terns.

SPA	Location	Pairs	Year designated	Site Condition Monitoring*	Recent counts	Year	Reference
UK North Sea & Channel							
Cromarty Firth	N Scotland	294 (1989-1993)	1999	Declined 2000	16 82 68	2008 2009 2010	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012
Inner Moray Firth	N Scotland	310	1999	No change 2000	0 0 0 0 0 0	2008 2009 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database SMP database SMP database

Ythan Estuary, Sands of Forvie	NE Scotland	265	1998	No change 2012	19 0 6 4	2004 2005 2006 2010	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012
Forth Islands	E Scotland	334 (1997-2001) Or 800 (Stroud et al. 2001)	1990 (and 2004)	Maintained 2003	191 155 197 17 26	2006 2007 2008 2010 2011	SMP database SMP database Lewis et al. 2012 Lewis et al. 2012 Stroud et al. 2014
Imperial Dock Lock	E Scotland	558	2004	Maintained 2009	989 789 732 818	2007 2008 2009 2010	SMP database Jennings 2012 Jennings 2012 Jennings 2012
Farne Islands	NE England	230 (1993-1997)	1985		118 117 104 98 112 101 88 94	2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Coquet Island	NE England	740 (1993-1997)	1985		1,226 1,228 1,022 1,228 1,358 1,193 1,158 1,041	2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
The Wash	E England	152 (1993)	1988		115 169 208 221	2010 2011 2012 2013	SMP database SMP database SMP database SMP database
North Norfolk Coast	E England	>460 (1996)	1989		434 437 347 270 198	2007 2008 2010 2011 2012	SMP database SMP database SMP database SMP database SMP database
Breydon Water		155 (1992-1996)	1996		197 181 170 173 158 93 92	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Foulness	S England	220 (1996)	1996		121 130 72 82 25	1998 2000 2002 2004 2008	SMP database SMP database SMP database SMP database SMP database

Dungeness to Pett Level	S England	266 (1993-1997)	1999		170 177 149 236 343 235 149 79	2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Poole Harbour (Brownsea Island)	S England	155 (1993-1997)	1999		248 157 180 185 191 222 171 163	2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Solent & Southampton Water	S England	267 (1993-1997)	1998		375 200 285 256 371 266 280	2002 2003 2004 2005 2006 2007 2007	SMP database SMP database SMP database SMP database SMP database SMP database Stroud et al. 2014
UK Western waters							
Glas Eileanan	W Scotland	530	1998	Maintained 2005	0 515 0 303 97 22	2006 2007 2008 2009 2011 2012	SMP database SMP database SMP database SMP database Lewis et al. 2012 SMP database
Carlingford Lough	N Ireland	339 (1993-1997)	1998		282 200 11 108 69 130 119	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Larne Lough	N Ireland	199 (1993-1997) Or 180 (Stroud et al. 2001)	1997		530 314 387 380 317 319 231	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Lough Neagh and Lough Beg	N Ireland	185 (1995)	1996		>54 >62 >73 >78	2010 2011 2012 2013	SMP database SMP database SMP database SMP database
Strangford Lough	N Ireland	603 (1993-1997)	1998		762 650 1,174 578 726 84 352	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database

The Dee Estuary	Engl-Wales	392 (1995-1999) Or 277 (Stroud et al. 2001)	1985		136 221 196 202 200 165	2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database
Ribble and Alt Estuaries		182 (1996)	1995		100 137 106 98 111 111	1997 1998 1999 2000 2003 2008	SMP database SMP database SMP database SMP database SMP database SMP database
Ynys Feurig, Cemlyn Bay	Wales	>189 (1992-1996)	1992		180 180 167 170 196 178 592	2005 2006 2007 2008 2009 2011 2011	SMP database SMP database SMP database SMP database SMP database SMP database Stroud et al. 2014

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

18.10 BDMPS

UK waters can be divided into two spatial BDMPS (UK North Sea and Channel, and UK western waters) which are appropriate for the migration seasons of this species (late July to early September, and April-May). This division into two BDMPS is based on the tendency for birds from UK colonies to migrate south after breeding and north back to their colony predominantly through the North Sea if birds breed at colonies in UK North Sea waters, or through UK western waters if birds breed at colonies in UK western waters, and for birds from European continental countries to migrate predominantly through UK North Sea waters rather than UK western waters. However, the common tern shows a greater tendency to migrate overland than seen in most other seabird species, so that use of a single BDMPS for all UK waters would also be a reasonable approach for this species.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 56 and 57.

Based on evidence reviewed in sections 18.5, 18.6 and 18.7, the UK North Sea and Channel migration seasons BDMPS is estimated to hold 70% of adults and 50% of immatures from UK North Sea colonies, 10% of adults and immatures from UK western waters colonies, 30% of birds from Norway, Finland, Sweden, Baltic States, 25% of birds from Germany and The Netherlands, 20% of birds from Ireland (Appendix A Table 56). These proportions give an estimated BDMPS of 144,911 birds, 18,942 from UK and 125,969 from overseas.

Based on evidence reviewed in sections 18.5, 18.6 and 18.7, the UK western waters migration seasons BDMPS is estimated to hold 30% of adults and 20% of immatures from UK North Sea colonies, 90% of adults and 60% of immatures from UK western waters colonies, 20% of birds from Norway, 10% of birds from Finland, Sweden, Baltic States and Germany, 5% of birds from The Netherlands, and 40% of birds from Ireland (Appendix A Table 57). These proportions give an estimated BDMPS of 64,659 birds, 16,212 from UK and 48,447 from overseas.

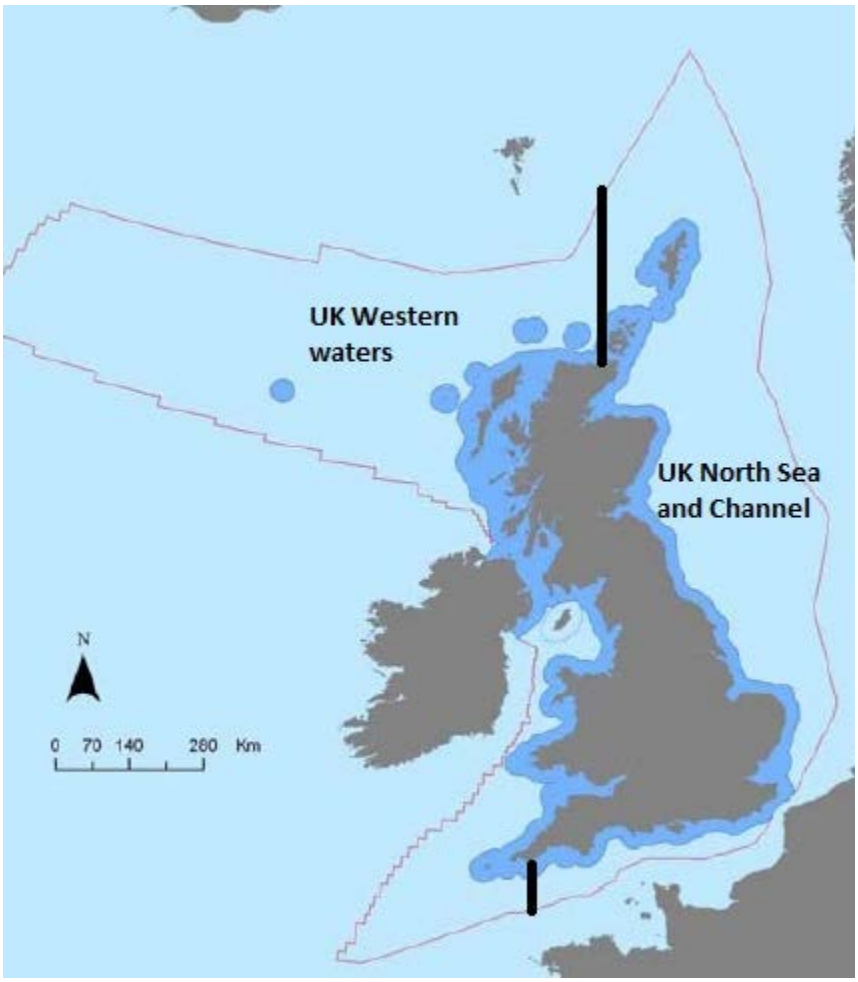


Figure 18.10. Two defined BDMPS spatial areas for common tern: 'UK North sea waters and Channel' and 'UK Western waters'.

18.11 Proportions of UK SPA birds in BDMPS

UK SPAs for common tern are widely distributed across the breeding range of the species in the UK. SPA birds represent about 44% of the UK population, so the main factor determining the proportion of each BDMPS derived from UK SPAs will be the ratio of overseas to UK birds in each BDMPS during the migration season. These percentages depend very much on the estimate of proportions of overseas populations migrating through UK waters so are very tentative estimates, as numbers of birds from overseas populations migrating through UK waters are very uncertain, although clearly are large. Proportions of birds that are adults from UK SPA colonies can be estimated directly from the data in Appendix A Tables 56 and 57. For example, in the UK western waters BDMPS (64,659 birds) there are estimated to be 4,126 adults from SPA colonies, so these represent 6% of the total birds present.

18.12 Spatial distribution of UK breeding SPA birds across the BDMPS

With large numbers of birds migrating through UK waters, and apparently many more overseas birds than UK birds in these migrations, the SPA birds are likely to be well mixed across each of the BDMPS areas.

19. ARCTIC TERN *Sterna paradisaea*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (July to early September, and late April to May) (adults and immatures)
Overseas	470,000	99,780
UK	158,000	135,548
Total	628,000	235,328

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Migration seasons BDMPS (July to early September, and late April to May)			
UK North Sea and Channel	163,930	82,084	81,846
UK Western waters	71,398	17,696	53,702

Arctic tern numbers in most UK SPA colonies are monitored frequently. However, numbers in UK colonies that are not SPA populations are less well monitored, and do represent a substantial proportion of the UK total. Arctic tern breeding numbers in SPA populations in the UK have declined very considerably, especially in Shetland and most of Orkney. How much numbers in UK non-SPA colonies have declined is far less clear, but numbers may be smaller than in the summary table above if non-SPA colonies have also declined as much as SPA colonies. Arctic tern migrations have not been studied by geolocator deployment except in Iceland (a population that does not pass through UK waters), and ringing recoveries from the migration period in UK waters are very limited. So understanding of details of Arctic tern movements is relatively poor, especially to the extent that birds from overseas populations are concerned. While ring recoveries show that many birds from overseas pass through UK waters, the proportions of those populations doing so are very uncertain since ring recovery data are subject to considerable potential bias. Furthermore, these overseas populations are large, and probably represent a moderate to high proportion of the total of Arctic terns in UK waters during the migration season. Therefore, estimated numbers of birds in the BDMPS are classed as red for the total population and numbers from overseas, and for numbers from the UK population given the uncertainty about breeding numbers in non-SPA colonies at present.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 58 and 59.

19.1 Breeding range and taxa

Arctic tern is monotypic, with a Holarctic breeding distribution, predominantly in Arctic and sub-Arctic regions. There appears to have been no assessment of whether biometrics would allow origins of individuals to be identified, but this seems unlikely as there seems to be no evidence of clinal variation, and birds are known to sometimes recruit to breed in locations far from their natal area.

19.2 Non-breeding component of the population

Arctic terns start to breed when 4 years old (BTO Birdfacts). Adult survival rate is 0.9 (BTO Birdfacts), juvenile survival unknown (BTO Birdfacts) and mean productivity is 0.402 chicks per pair (JNCC database, n=227 measurements). To obtain a stable population, survival of immatures was adjusted to 0.72 for juveniles, 0.85 for 1-year olds, and 0.9 for 2-year olds and 3-year olds. The model population comprised 63% adults, 13% juveniles and 24% older immatures. There are 0.58 immatures per adult.

19.3 Phenology

Breeding colonies in the UK are deserted by mid-August, with modal departure in late July or early August (Pennington et al. 2004; Forrester et al. 2007). Autumn dispersal/migration starts in early July (Pennington et al. 2004; Forrester et al. 2007) or late July (Cramp et al. 1977-94). Peak autumn migration occurs in late July in Shetland (Pennington et al. 2004) and Scotland (Forrester et al. 2007), but continues from August to October when considering the entire migration to Antarctic waters (Cramp et al. 1977-94). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late July and early August (Figure 19.1). Autumn migration is completed in Shetland by late August (Pennington et al. 2004) and in Scotland and England by September (Brown and Grice 2005; Forrester et al. 2007), but in the southern hemisphere may continue until mid-November (Cramp et al. 1977-94).

Spring migration starts in the southern hemisphere in early March (Cramp et al. 1977-94), and the first migrants appear in UK waters in March (Wernham et al. 2002), but in Scottish waters and Shetland not until late April (Pennington et al. 2004; Forrester et al. 2007). Peak spring migration occurs in mid-May in UK waters (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in early May (Figure 19.1). Spring migration is completed by late May (Pennington et al. 2004), early June (Cramp et al. 1977-94), or June (Forrester et al. 2007).

The first spring records of Arctic tern in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 6 April to 9 May but mostly in late April, and the last records were from 9 September to 21 November, but mostly in late October. Peak autumn migration was reported in July or July-August in most years, and peak spring migration was reported in May in almost all years. Birds re-occupy colonies from late April, with modal return in mid-May (Pennington et al. 2004; Forrester et al. 2007).

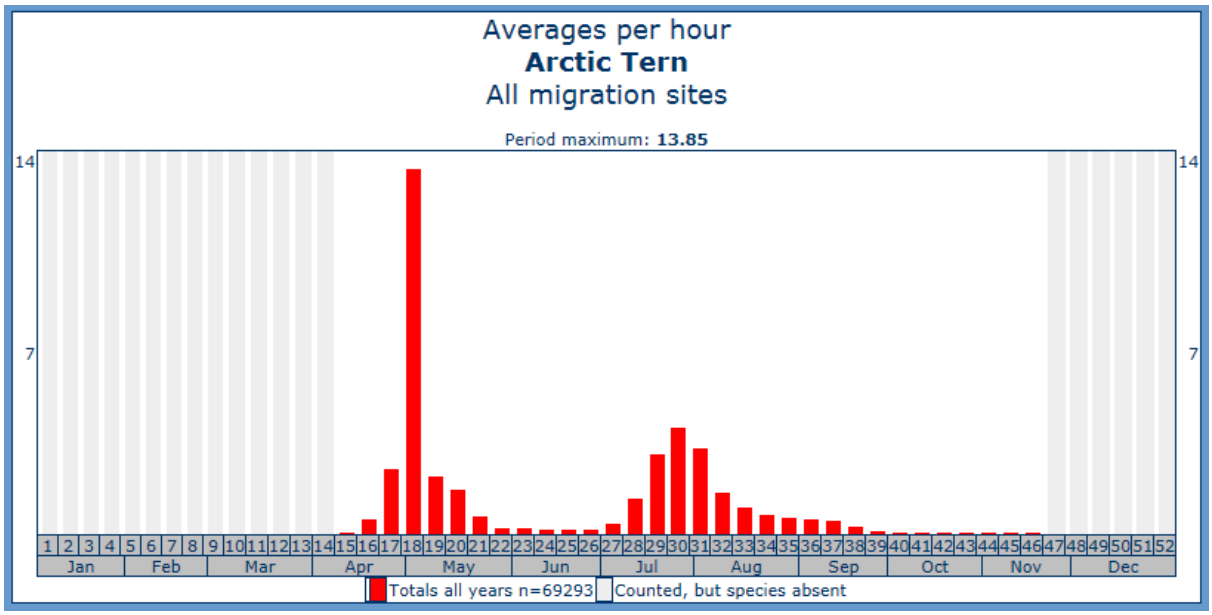


Figure 19.1. Average numbers of Arctic terns counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season birds predominantly absent from UK waters. However, from the data reviewed above, this may be refined to breeding season May-early August, non-breeding season mid-August-April.

19.4 Defined seasons:

- UK Breeding season — May-early August
- Post-breeding migration in UK waters — July-early September (**migration BDMPS**)
- non-breeding season — mid August-April
- Return migration through UK waters late — April-May (**migration BDMPS**)
- Migration-free breeding season — June
- Migration-free winter season — October-March

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for Arctic tern:

Migration periods BDMPS (July-early September, and late April-May).

19.5 Movements of birds from the UK population

Although Post-breeding dispersal occurs from colonies in July (with some UK fledglings moving as far as the Baltic Sea), followed by southwards migration in August-September Wernham et al. 2002; Pennington et al. 2004; Forrester et al. 2007). Several Scandinavian and Baltic fledglings have also been recovered in the UK as early as August, indicating rapid dispersal of some young birds into UK waters. As with other terns, post-fledging dispersal takes birds to areas with high density of prey fish where juveniles have a good chance of learning fishing skills, surviving and putting on weight before the southwards migration. In contrast to many other terns, Arctic terns seem less likely to remain in family groups and chicks seem to become independent rather quickly. Arctic terns are thought to migrate somewhat further offshore than other British tern species, past west Africa to southern Africa then onwards to the edge of Antarctic pack ice (Wernham et al. 2002). Movements of first summer and second summer birds are not well documented, but it appears that most first

and second summer birds remain in the southern hemisphere all year, with very few of these birds in immature plumage returning to UK waters in summer (Wernham et al. 2002). Some three year olds breed, while others visit breeding areas to loaf at 'club' sites on the periphery of the colony. Most four year olds breed, but it is likely that many recruit into colonies away from where they were reared, while there is also some evidence for adults moving colony between years (Wernham et al. 2002).

19.6 Movements of birds from overseas into UK waters

Foreign-ringed birds recovered in UK waters, or on shore, mainly originate from Scandinavia and the Baltic (Wernham et al. 2002). Post-breeding dispersal/migration can be very rapid and can bring birds from overseas populations into close contact with local breeding populations in late summer. For example, a bird ringed as a chick in the Baltic States in early July was killed by a great skua hunting for terns roosting adjacent to the Arctic tern colony at Foula in mid-July, just a few days after it had fledged; without the ring this bird would have been assumed to be a local fledgling from the Foula colony. Although there is one recovery in the UK of a chick ringed in Greenland, no birds ringed in Iceland have been found in the UK according to Wernham et al. (2002). However, three out of over 12,000 ringed in Faroe (88% as chicks) were recovered during autumn migration in the British Isles (Hammer et al. 2013). Seabird 2000 reported 53,380 pairs in UK, 2,730 in Ireland, 131,000 pairs in Fennoscandia, 8,000 pairs in the Baltic States, 375,000 pairs in Iceland (Mitchell et al. 2004), and Hammer et al. (2013) report 7,600 pairs in Faroe. Given the evidence for extensive post-breeding dispersal of birds from Fennoscandia and the Baltic into UK waters, and the large populations in those areas, it seems likely that a substantial proportion of Arctic terns in UK waters in August-September will be from those regions. Recent breeding failures of Arctic terns in Iceland, and circumstantial observational evidence at colonies, suggest that numbers there may well have declined considerably (Vigfusdottir et al. 2013). Given the very large size of the Icelandic population, those birds might be expected to form a substantial part of the total in UK waters in August-September. However, deployment of geolocators on ten Arctic terns in Greenland and one in Iceland showed that all eleven birds moved directly south from Iceland to the Newfoundland Basin, where they spent some time before migrating to the South Atlantic (Egevang et al. 2010). All birds showed essentially the same route, with none coming near to UK waters. Return migration in spring was even further to the west, passing close to Newfoundland before completing the journey to Iceland and Greenland. This study suggests that very few Arctic terns from Iceland and Greenland ever visit UK waters, consistent with the lack of recoveries of Arctic terns ringing in Iceland in the British Isles. Spring migration through UK waters (some of which can occur overland; Wernham et al. 2002) may also involve large numbers from colonies in Fennoscandia and the Baltic, but the spring migration produces few ring recoveries so this is uncertain.

19.7 Numbers in UK waters

Terns are very difficult to census during migrations. However, Forrester et al. (2007) suggest that there are 10,000 to 200,000 on passage through Scottish waters in autumn and spring. No equivalent estimates for other parts of UK waters appear to be published. Clearly all UK breeders, and probably about half of the immatures associated with these pass through UK waters during the migration seasons, but so do large numbers of birds from overseas. Numbers from those populations passing through are very uncertain.

19.8 Biogeographic population and relevant smaller units (BDMPS)

Stroud et al. (2001) defined the biogeographic breeding population as that of the European and North Atlantic population, comprising 900,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 493,000-1,800,000 pairs. Kober et al. (2010) did not present an estimated biogeographic population for this species. Populations with possible connectivity to UK waters are the UK population (50,000 pairs), and the populations of Fennoscandia (131,000 pairs), Faroe (7,600 pairs), Baltic states (8,000 pairs)

and Ireland (2,500 pairs) (Figure 19.2). So overseas populations are large relative to the UK population, but the proportion of these overseas birds that pass through UK waters is very uncertain. The biogeographic population with connectivity to UK waters is estimated at 628,000 birds (adults and immatures) with 158,000 from UK and 470,000 from overseas populations. Allowing for the likely proportions of each population that pass through UK waters on migration, the total numbers in UK waters during the migration seasons is estimated at 236,000 birds (adults and immatures) with 136,000 of these from the UK population and 100,000 from overseas.

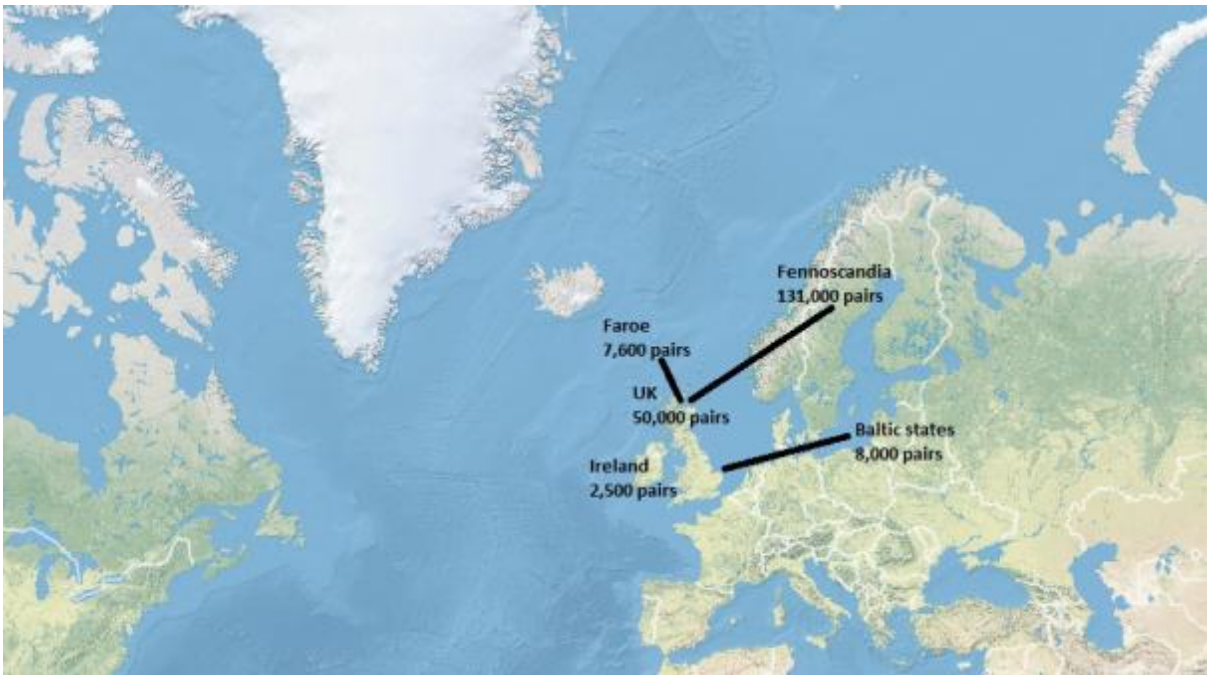


Figure 19.2. Breeding population origins of Arctic terns in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

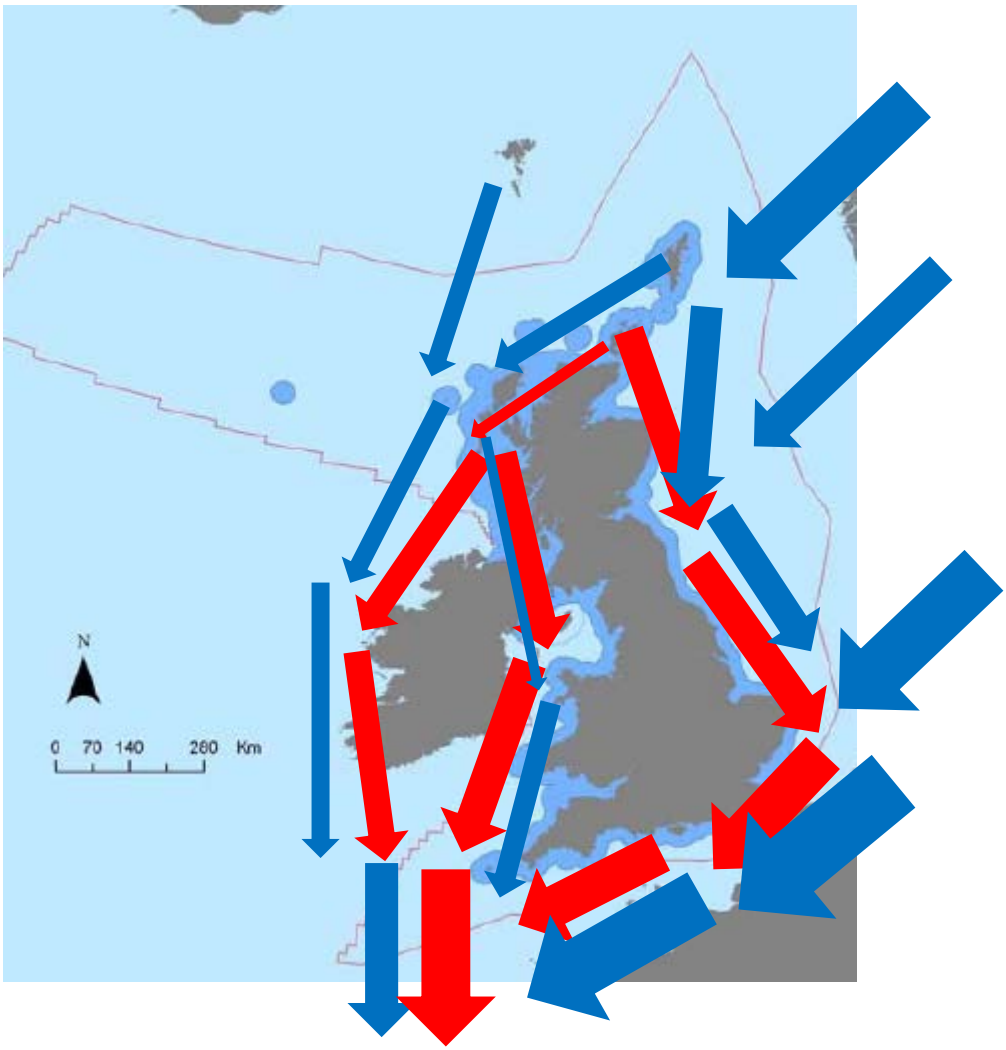


Figure 19.3. Main movements of Arctic terns from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply specific overland migration routes, although this species may sometimes migrate over land. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

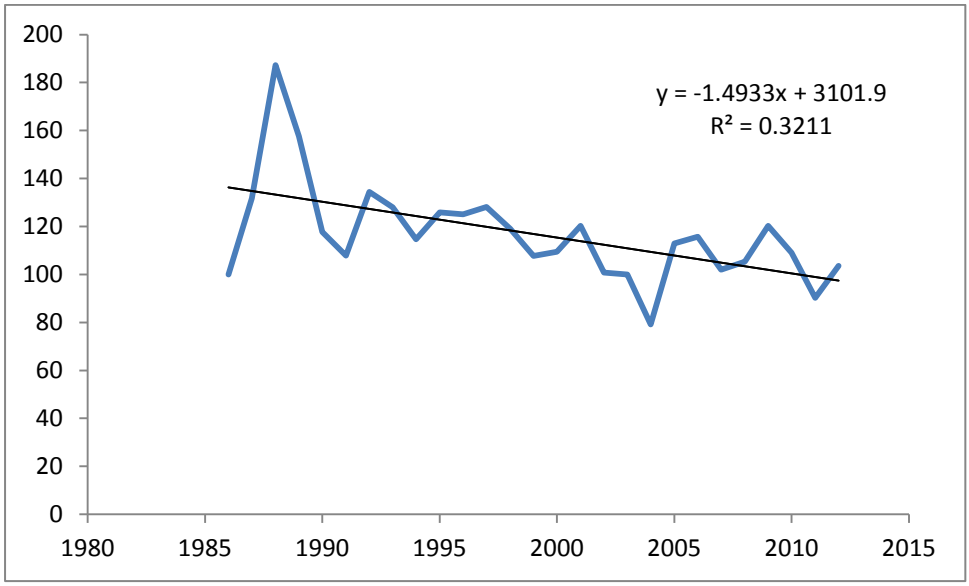


Figure 19.4. Trend in the Arctic tern breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

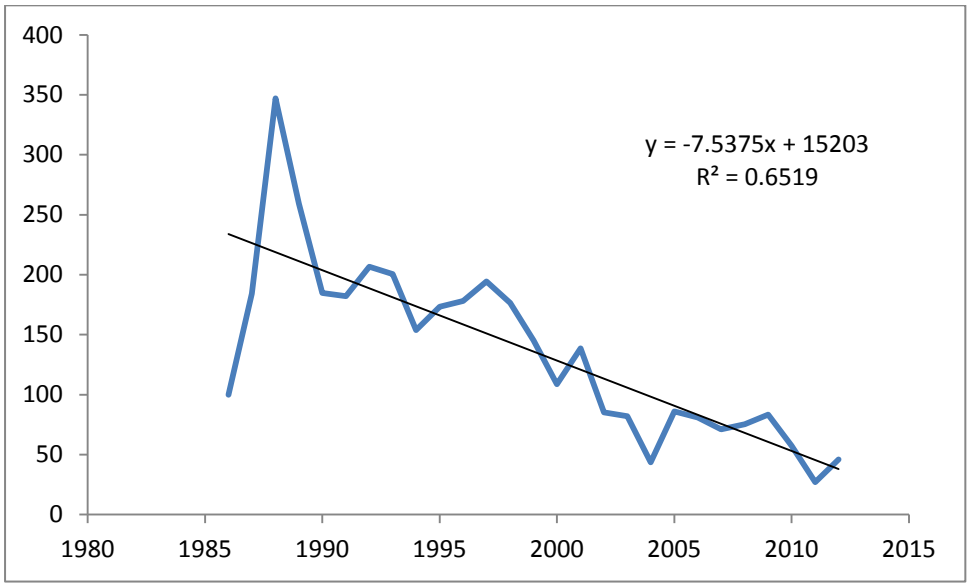


Figure 19.5. Trend in the Arctic tern breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

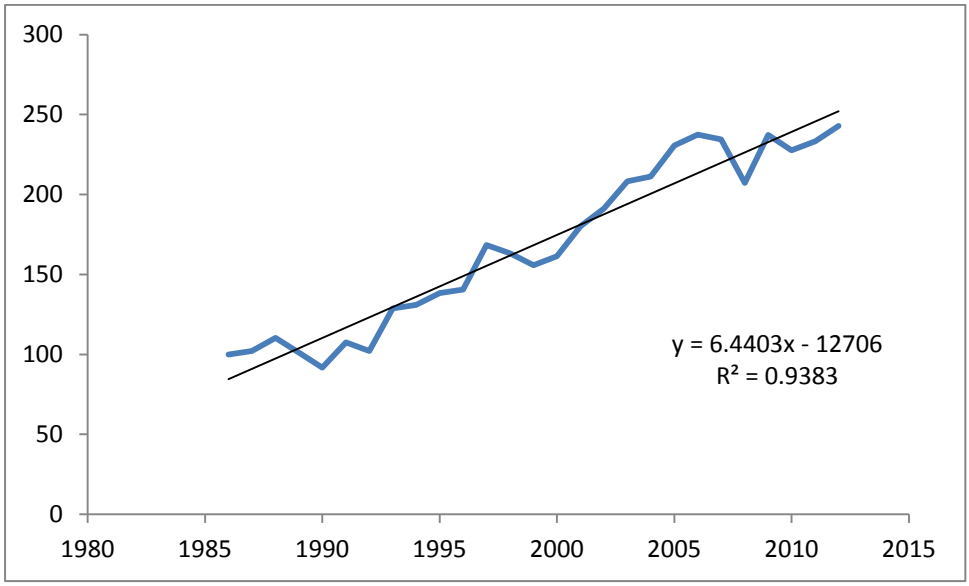


Figure 19.6. Trend in the Arctic tern breeding population index in England from 1986-2012. Data from JNCC seabird population monitoring database.

19.9 Proportion of UK population from UK breeding SPAs

The 17 SPAs with breeding Arctic terns as a feature together held 17,124 pairs at designation, estimated to represent ca. 38% of the British breeding population and 17% of the all-Ireland breeding population (Stroud et al. 2001). Stroud et al. (2014) suggest on the basis of census data at these sites in 2000-2011 that 20.6% of the GB breeding population is on UK SPAs for breeding Arctic terns. This decrease is consistent with a density-dependent effect of food shortage, reducing breeding numbers proportionately more at larger colonies, which is very likely to occur and has been shown in several other seabird species although not specifically for Arctic tern.



Figure 19.7. The UK SPA suite for breeding Arctic terns. These SPA populations are listed in Table 19.1.

Table 19.1. The UK SPA suite for breeding Arctic terns.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
Fetlar	Shetland	520 (1994-1997)	1994	Recovering 2002	486 213 16 14 2 0 21	2006 2007 2008 2009 2010 2011 2012	SMP database SMP database Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database SMP database
Foula	Shetland	1,100 (1992-1997)	1995	Maintained 2000	0 70 35 100 20	2006 2007 2011 2012 2013	Lewis et al. 2012 SMP database SMP database Gear 2012 Gear 2013
Papa Stour	Shetland	1,000	2000	Declined 2008	1,172	2000	Stroud et al. 2014

Mousa	Shetland	767 (1994)	1995	No change 2000	143 751 400 925 42 0 41 18	2001-06 2007 2008 2009 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database SMP database SMP database
Sumburgh Head	Shetland	700 (1994)	1996	Declined 2001	ca40 ca150 203	1999 2000 2000	SCM database SCM database Stroud et al. 2014
Fair Isle	Shetland	1,120 (1993-1997)	1994	Declined 2009	818 208 0 283 400 9 227 29	2006 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database FIBO Report SMP database
West Westray	Orkney	1,200	1996	Declined 2007	1,067 ca500	2000 2009	Stroud et al. 2014 SCM database
Papa Westray	Orkney	1,950	1996	Declined 2006	813 556 393 176	2005 2006 2010 2011	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Stroud et al. 2014
Rousay	Orkney	1,000	2000	Declined 2007	707 ca60	2000 2006	Stroud et al. 2014 SCM database
Auskerry	Orkney	780 (1995)	1998	Maintained 2007	0 550 667 0 750	2005 2006 2007 2011 2013	SMP database SMP database SMP database Lewis et al. 2012 SMP database
Pentland Firth Islands	N Scotland	1,200 (1992-1995)	1997	Declined 2007	327 1,400 0 669	2004 2005 2007 2009	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Stroud et al. 2014
Forth Islands	E Scotland	540 (1992-1996)	1990	Declined 2009	515 525 511 316 34 250 265	2006 2007 2008 2009 2010 2011 2012	SMP database SMP database SMP database Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SMP database
Farne Islands	NE England	2,840 (1993-1997)	1985		2,256 2,239 2,198 2,199 1,830 1,866 1,921	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database

Coquet Island	NE England	700 (1993-1997)	1985		1,247 983 1,259 1,046 1,140 1,275 1,224	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database
UK Western waters							
Outer Ards	N Ireland	207 (not stated)	2002		182 215 191 174 108 60	2008 2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database SCM database
Strangford Lough	N Ireland	210 (1993-1997)	1998		891 559 316 645 373 229 55 164	2006 2007 2008 2009 2010 2011 2012 2013	SCM database SCM database SCM database SCM database SCM database SCM database SCM database SCM database
Ynys Feurig, Cemlyn Bay	Wales	1,290 (1992-1996)	1992		540 493 416 531 550 3,620	2006 2007 2008 2009 2011 2011	SMP database SMP database SMP database SMP database SMP database Stroud et al. 2014

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

19.10 BDMPS

The UK waters can be divided into two spatial BDMPS; UK North Sea and Channel waters, and UK western waters (Figure 19.8), which are appropriate for the migration seasons of this species (July to early September, and late-April to May). This division into two BDMPS is based on the tendency for birds from UK colonies to migrate south after breeding and north back to their colony predominantly through the North Sea if birds breed at colonies in UK North Sea waters, or through UK western waters if birds breed at colonies in UK western waters, and for birds from European continental countries to migrate predominantly through UK North Sea waters rather than UK western waters.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 58 and 59.

Based on evidence reviewed in sections 19.5, 19.6 and 19.7, the UK North Sea and Channel migration seasons BDMPS is estimated to hold 90% of adult and 60% of immature Arctic terns from Orkney and Shetland colonies, 100% of adults and 70% of immatures from colonies along the mainland east coast of Scotland and England, none from colonies in UK western waters, 20% of adults and 15% of immatures from Fennoscandia and Faroe, 10% of birds from the Baltic States, but none from Ireland (Appendix A Table 58). These proportions lead to an estimated BDMPS of 163,930 birds, 81,846 from the UK and 82,084 from overseas populations.

Based on evidence reviewed in sections 19.5, 19.6 and 19.7, the UK western waters migration seasons BDMPS is estimated to hold 10% of adult and 10% of immature Arctic terns from Orkney and Shetland colonies, no adults but 10% of immatures from colonies along the mainland east coast of Scotland and England, 100% of adults and 70% of immatures from colonies in UK western waters, 3% of adults and immatures from Fennoscandia, 10% of adults and immatures from Faroe, 2% of birds from the Baltic States, and 30% of birds from Ireland (Appendix A Table 59). These proportions lead to an estimated BDMPS of 71,398 birds, 53,702 from the UK and 17,696 from overseas populations.

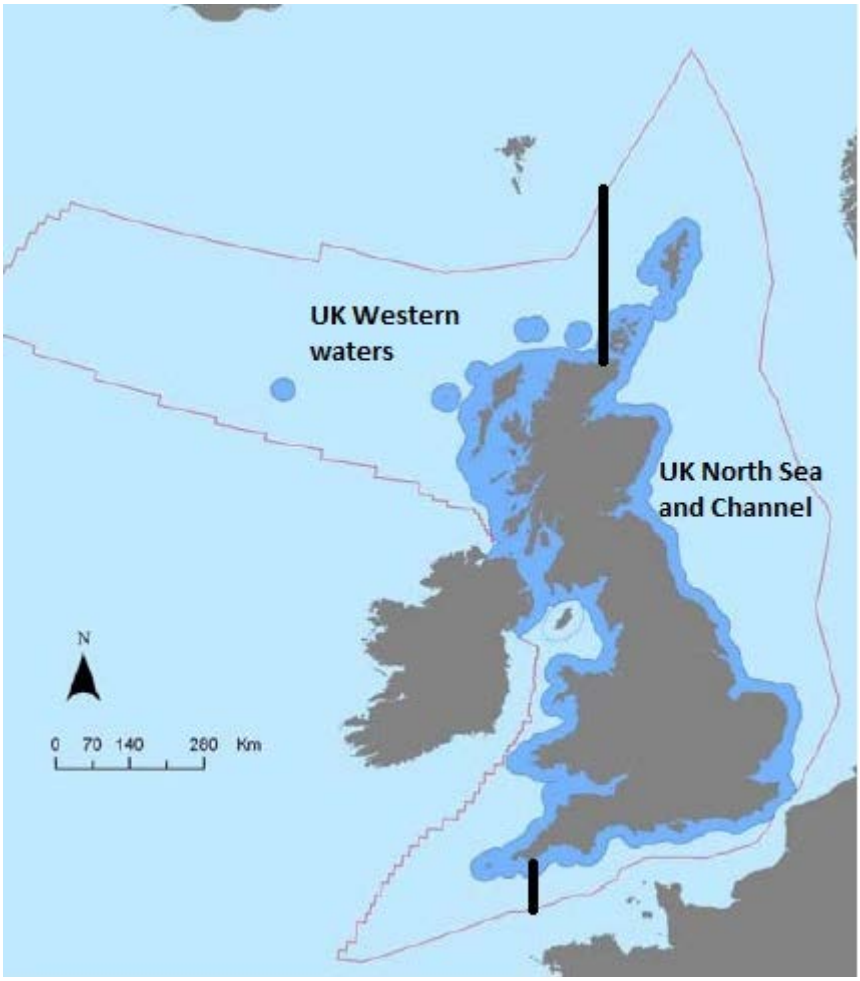


Figure 19.8. Two defined BDMPS spatial areas for Arctic tern: 'UK North Sea waters and Channel'and 'UK Western waters'.

19.11 Proportions of UK SPA birds in BDMPS

UK SPAs for Arctic tern are strongly concentrated in Shetland and Orkney, but with little representation in the west of Scotland. Numbers breeding in Shetland and Orkney have declined very considerably since the 1980s, at least in part as a result of declines in sandeels in the NW North Sea. SPA birds probably represent about 20% of the UK population now (section 19.9), so the main factor determining the proportion of each BDMPS derived from UK SPAs will be the ratio of overseas to UK birds in each of the two BDMPS during the migration season. These percentages depend very much on the estimate of proportions of overseas populations migrating through UK waters so are very tentative estimates, as numbers of birds from overseas populations migrating through UK waters are very uncertain. Proportions of birds that are adults from UK SPA colonies can be estimated directly from the data in Appendix A Tables 58 and 59. For example, in the UK western

waters BDMPS (71,398 birds) there are estimated to be 2,138 adults from SPA colonies, so these represent 3% of the total birds present.

19.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Since birds can disperse quickly from colonies, but may stop to feed at locations where there are suitable food stocks, UK SPA birds are likely to be well mixed among non-SPA populations and overseas populations also passing through and responding to the same opportunities.

20. LITTLE TERN *Sternula albifrons*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (late July to early September, and mid-April to May) (adults and immatures)
Overseas	620	514
UK	5,620	4,612
Total	6,240	5,126

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Migration seasons BDMPS (late July to early September, and mid-April to May)			
UK North Sea and Channel	3,524	0	3,524
UK Western waters	1,602	514	1,088

Little tern breeding numbers are well monitored at most SPA colonies, and the SPA colonies hold a fairly high proportion of the total UK population of this species. The only overseas population of little terns to migrate through UK waters is the Irish population, and it seems almost certain that almost all adult little terns from the well-studied population in Ireland pass through UK waters in SW Approaches during migration. Therefore, all categories are coded green.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 60 and 61.

20.1 Breeding range and taxa

Little tern has a wide breeding range that includes the Palearctic, Afrotropic and Australasian regions. There are six subspecies, but only the nominate *S. a. albifrons* occurs in British waters. That subspecies breeds across most of Europe (but not in northern areas and with largest numbers mainly in southern countries) to central Asia and northern India, and in North Africa. There appears to have been no assessment of whether biometrics would allow origins of individuals to be identified.

20.2 Non-breeding component of the population

Little terns start to breed when 3 years old (BTO Birdfacts). Adult survival rate is 0.899 (BTO Birdfacts), juvenile survival 0.578 (BTO Birdfacts) and mean productivity is 0.521 chicks per pair (JNCC database, n=362 measurements). To obtain a stable population, survival of immatures was adjusted to 0.65 for juveniles, 0.75 for 1-year olds, and 0.8 for 2-year olds. The model population comprised 64% adults, 17% juveniles and 19% older immatures. There are 0.56 immatures per adult.

20.3 Phenology

Breeding colonies in the UK are deserted by August, with modal departure in late July (Forrester et al. 2007). Autumn migration starts in mid-July (Cramp et al. 1977-94) or late July (Forrester et al. 2007). Peak autumn migration occurs in August (Wernham et al. 2002), August-September (Forrester et al. 2007), or August-October considering the entire range in Europe (Cramp et al. 1977-94). Peak numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late July with quite rapid decrease in numbers through August (Figure 20.1). Autumn migration is completed by September (Wernham et al. 2002), early October in Scotland (Forrester et al. 2007), mid-October in England (Brown and Grice 2005).

Spring migration starts in March in southern Europe (Cramp et al. 1977-94), but first migrants arrive in UK waters in April (Wernham et al. 2002) and in mid-April in Scottish waters (Forrester et al. 2007). Peak spring migration occurs in mid-April to mid-May in English waters (Brown and Grice 2005), late April in Scotland (Forrester et al. 2007) or April-May (Cramp et al. 1977-94; Wernham et al. 2002). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in very late April and early May (Figure 20.1). Spring migration is completed by May (Forrester et al. 2007) or late May (Cramp et al. 1977-94).

The first spring records of little tern in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from 9 April to 12 May, but mostly in mid- to late-April, and the last records were from 21 July to 29 September, but mostly in early August. Peak autumn migration was reported in July in most years, and peak spring migration was reported in late April or in May in most years. Birds re-occupy colonies from mid-April, with modal return in late April (Forrester et al. 2007).

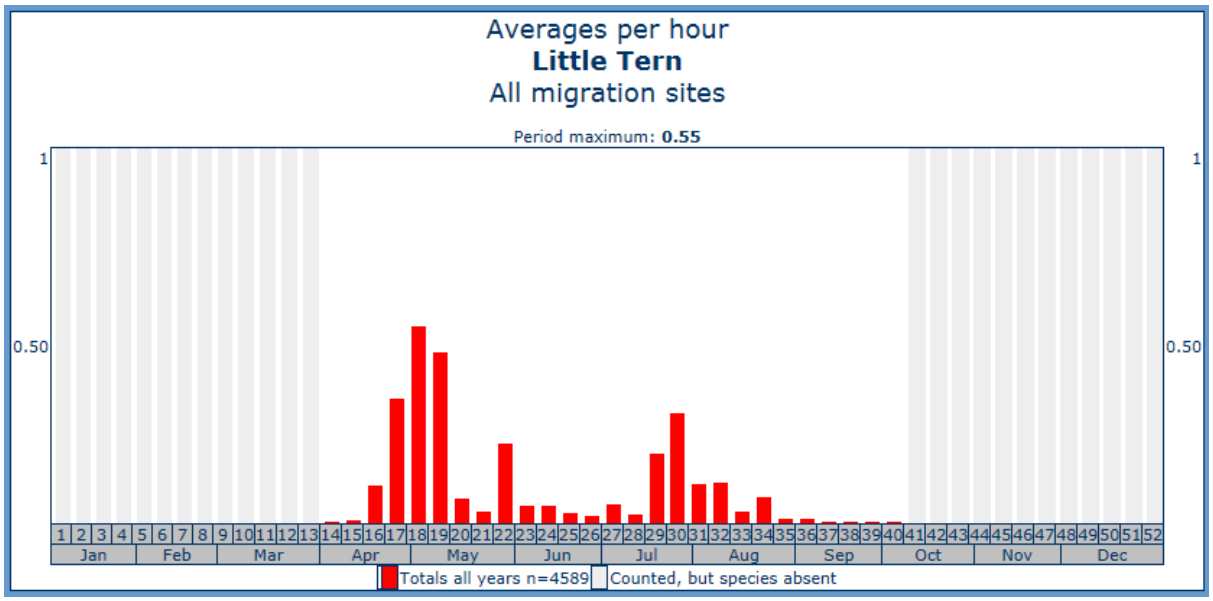


Figure 20.1. Average numbers of little terns counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-August, non-breeding season birds predominantly absent from UK waters. However, from the data reviewed above, this may be refined to breeding season May-early August, non-breeding season mid August-April.

20.4 Defined seasons:

- UK Breeding season — May-early August
- Post-breeding migration in UK waters — late July-early September (**migration BDMPS**)
- non-breeding season — mid August-April
- Return migration through UK waters — mid April-May (**migration BDMPS**)
- Migration-free breeding season — June
- Migration-free winter season — October-March

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for little tern:

Migration periods BDMPS (late July-early September, and mid April-May).

20.5 Movements of birds from the UK population

Birds depart rather rapidly after the breeding season, with ring recoveries from southern Europe as early as August, one within 6 days of ringing at a colony in England (Wernham et al. 2002). Large flocks of little terns in The Netherlands in August suggest that is a staging area used by birds from a wide geographical area during autumn migration (Wernham et al. 2002). Several ring recoveries of birds from Scottish colonies have been in Denmark, whereas most English birds have been recovered in The Netherlands, suggesting that Scottish birds cross the North Sea eastwards from Scotland rather than flying southwards (Wernham et al. 2002). Little terns do not breed until they are at least two years old, and it has been assumed that they spend their first year in African winter quarters, but there is no ringing evidence to support this (Wernham et al. 2002).

20.6 Movements of birds from overseas into UK waters

Wernham et al. (2002) point out that we know little about whether there is passage through UK waters of birds breeding elsewhere. Presumably at least the Irish population (210 pairs in Seabird 2000; Mitchell et al. 2004) must pass through UK waters on migration between Ireland and Africa, but while there are quite large numbers in Fennoscandia (1,019 pairs), the Baltic States (550 pairs), Germany (870 pairs), The Netherlands (500 pairs) and Belgium (224 pairs) (Mitchell et al. 2004) there is no evidence that any of these birds cross the North Sea into UK waters, while ring recovery data suggest that they do not, but that tose populations migrate through continental Europe.

20.7 Numbers in UK waters

Direct observation gives no indication of numbers passing through UK waters, as little terns seem rarely to be observed except in the immediate vicinity of colonies (see for example how few are recorded in the Trektellen data set for UK waters, Figure 20.1). Forrester et al. (2007) refrain from suggesting how many pass through Scottish waters, but comment 'little tern is rare outside its breeding range'. Nevertheless, it is clear that UK and Irish little terns must migrate through UK waters, while it seems that no birds from other populations do so. Therefore, numbers can be estimated from population sizes, which are fairly accurately known.

20.8 Biogeographic population and relevant smaller units (BDMPS)

Stroud et al. (2001) defined the biogeographic breeding population as that of the European population, comprising 20,643 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 17,000-22,000 pairs. Kober et al. (2010) did not present an estimated biogeographic population for this species. In terms of populations with connectivity to UK waters it would appear that only the UK population (1,800 pairs) and Irish population (200 pairs) are likely to migrate through UK waters (Figure 20.2). These 3,600 UK adults will have an associated 2,000 or so immatures, but perhaps half of these may not migrate into UK waters as young immatures, so the UK population in UK waters during the migration seasons may be around 4,600 birds (slightly more in autumn and fewer in spring). The 200 pairs from Ireland will similarly have associated immatures, giving a total population that may migrate through UK waters of about 500 birds. Thus the biogeographic population with connectivity to UK waters is estimated at 6,240 birds (adults and immatures), with 5,620 from UK and 620 from overseas. Of these, it is estimated that 5,120 birds migrate through UK waters, with 4,610 being from the UK population and 510 from overseas. The number from the UK population migrating through UK waters is less than the number contributing to the biogeographic population because it is believed that many first summer birds remain in their winter area rather than returning to the UK.



Figure 20.2. Breeding population origins of little terns in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

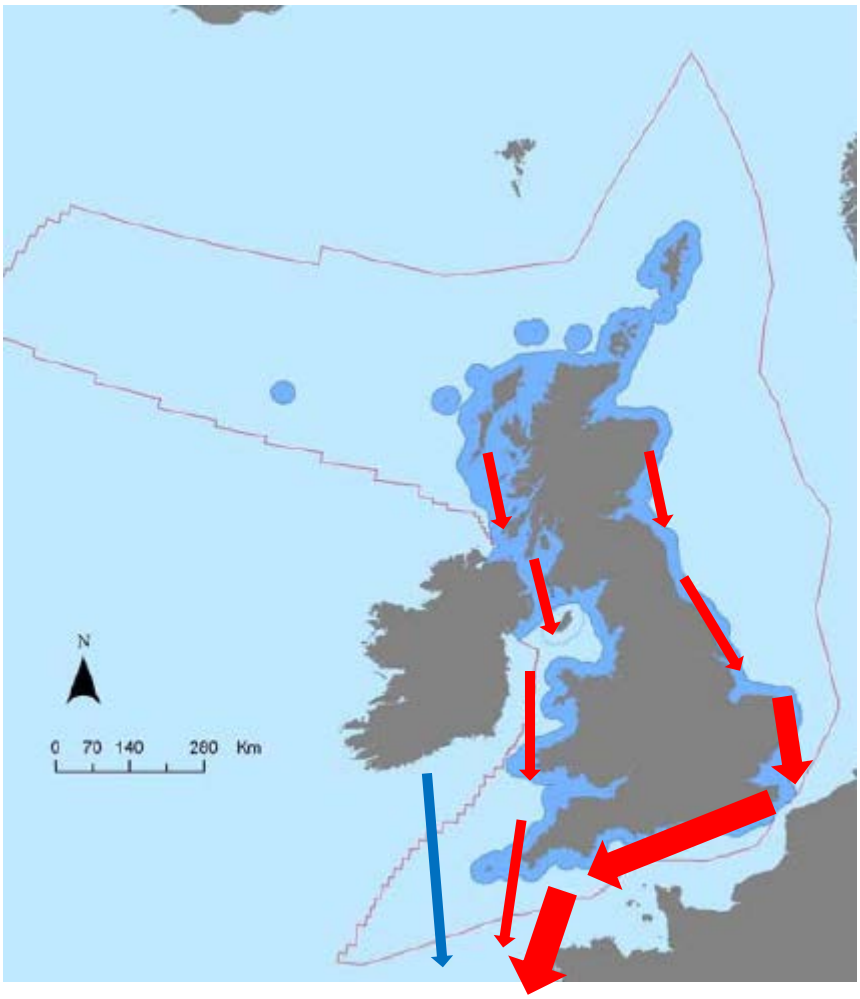


Figure 20.3. Main movements of little terns from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

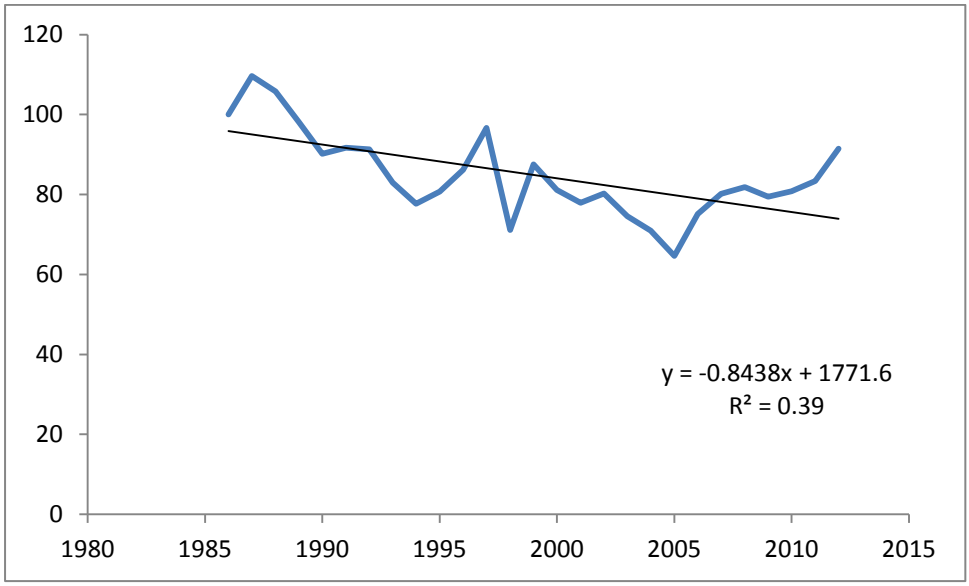


Figure 20.4. Trend in the little tern breeding population index in UK (which come almost entirely from colonies in England) from 1986-2012. Data from JNCC seabird population monitoring database.

20.9 Proportion of UK population in UK breeding SPAs

The 27 SPAs with breeding little terns as a feature together held 1,616 pairs at designation, estimated to represent ca. 67% of the British breeding population (Stroud et al. 2001). Stroud et al. (2014) suggest on the basis of census data for these populations from 2000-2011 (but mostly from 2011) that the GB SPA suite for breeding little terns held 61% of the GB population in that period. Numbers of little terns in the UK appear to have declined only slightly in recent years (Figure 20.4), but the decrease in proportion on SPAs suggests losses from some SPA populations have been greater than in the overall population.

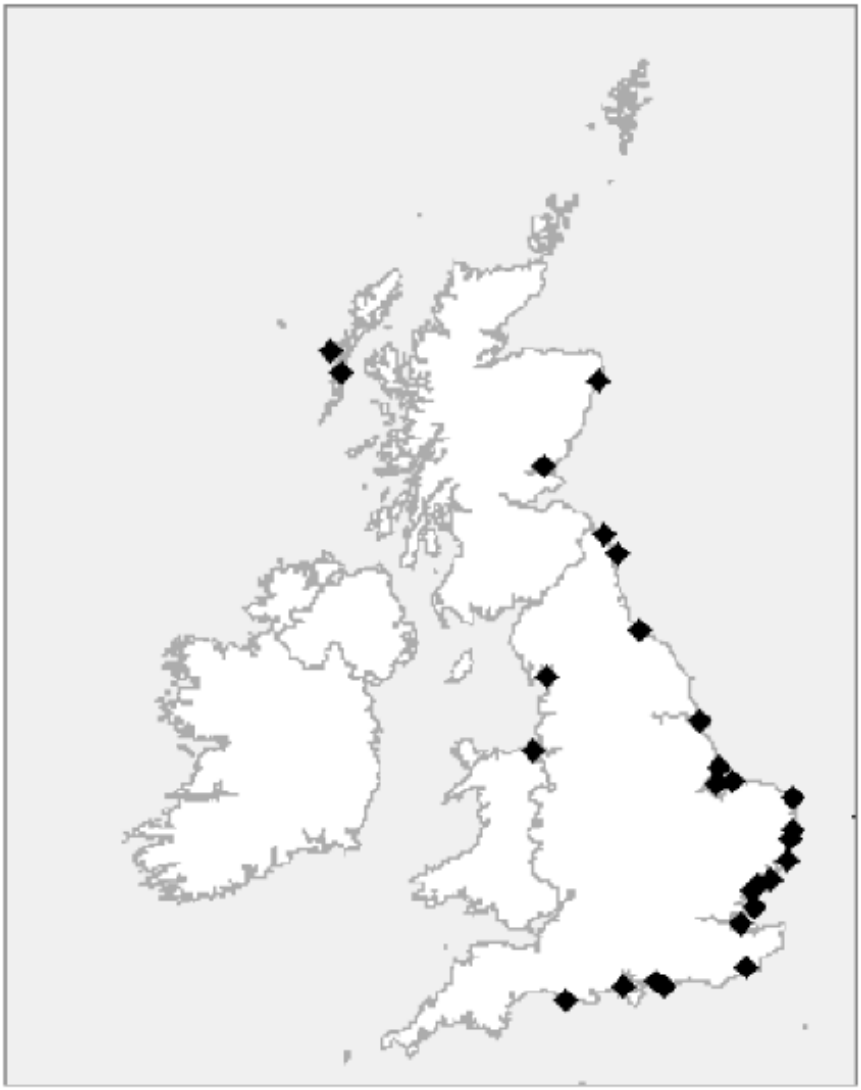


Figure 20.5. UK SPA suite for breeding little terns. These SPA populations are listed in Table 20.1.

Table 20.1. The UK SPA suite for breeding little terns.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count	Year	Reference
UK North Sea & Channel							
Ythan Estuary, Sands of Forvie	NE Scotland	41	1998	Maintained 2012	21 36 37 31 27 40	2008 2009 2010 2011 2012 2013	Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 Lewis et al. 2012 SCM database SCM database
Firth of Tay and Eden Estuary	E Scotland	44	2000	No change 2001	1 1	2005 2007	Lewis et al. 2012 Lewis et al. 2012

Lindisfarne	NE England	15 (1992-1996) Or 38 (Stroud et al. 2001)	1992		8	2011	Stroud et al. 2014
Northumbria Coast	NE England	40 (1992-1996)	2000		38	2011	Stroud et al. 2014
Teesmouth & Cleveland Estuary	NE England	40 (1995-1998) Or 37 (Stroud et al. 2001)	1995		84	2011	Stroud et al. 2014
Gibraltar point	Lincs	23 (1992-1996)	1993		12	2011	Stroud et al. 2014
Humber Flats, Marshes & Coast	E England	51 (1998-2002) Or 63 (Stroud et al. 2001)	2007		29	2011	Stroud et al. 2014
The Wash	E England	>33 (1992-1996)	1988		0	2009 - 2010	Stroud et al. 2014
North Norfolk Coast	E England	>330 (1992-1996) Or 377 (Stroud et al. 2001)	1989		409	2011	Stroud et al. 2014
Alde-Ore Estuary	E England	48 (1993-1997)	1996		0	2009	Stroud et al. 2014
Minsmere-Walberswick	E England	28 (1992-1996)	1992		30	2010	Stroud et al. 2014
Great Yarmouth North Denes	E England	220 (1992-1996)	1993		5	2011	Stroud et al. 2014
Foulness	Essex	>24 (1992-1996)	1996		0	2005	Stroud et al. 2014
Dungeness to Pett Level	SE England	35 (1993-1997)	1999		10 14 11	2011 2012 2013	SMP database SMP database SMP database
Medway Estuary and Marshes	Kent	28	1995		18	2009	Stroud et al. 2014
Benacre to Easton Bavents	E England	21 (1992-1996) Or 53 (Stroud et al. 2001)	1996		45	2011	Stroud et al. 2014

Blackwater Estuary	Essex	>21 (1992-1996) Or 36 (Stroud et al. 2001)	1995		99	2000	Stroud et al. 2014
Colne Estuary	Essex	>38 (1992-1996)	1994		0	2011	Stroud et al. 2014
Hamford Water	Essex	55 (1992-1995)	1993		45	2011	Stroud et al. 2014
Chesil Beach and The Fleet	S England	55	1985		19	2011	Stroud et al. 2014
Chichester & Langstone Harb	S England	100 (1992-1996)	1987		60	2011	Stroud et al. 2014
Pagham Harbour	Sussex	7 (1992-1996) Or 12 (Stroud et al. 2001)	1988		6	2011	Stroud et al. 2014
Solent & Southampton Water	S England	49 (1993-1997)	1998		0	2007	Stroud et al. 2014
UK Western waters							
Monach Isles	Western Isles	26 (1992)	1994	Declined 2001	2	2001	Seabird2000
South Uist Machair & Lochs	Western Isles	31 (1986-1990)	1997	Declined 2009	7 17	1999 2002	Seabird2000 Stroud et al. 2014
The Dee Estuary	Cheshire & Flintshire	69 (1995-1999) Or 56 (Stroud et al. 2001)	1985		126	2011	Stroud et al. 2014
Morecambe Bay	NW England	26	1996		62	2011	Stroud et al. 2014

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

20.10 BDMPS

UK waters can be split into two distinct spatial BDMPS which are appropriate for the migration periods (late July to early September, and mid-April to May): UK North Sea and Channel waters, and UK western waters. These are distinct for little tern because the evidence suggests that birds from colonies in the North Sea and Channel rarely migrate into UK western waters and vice versa. In addition, although birds from Ireland migrate through UK western waters so contribute to that BDMPS, no significant numbers of little terns from overseas populations are thought to migrate through UK North Sea waters. Numbers of this species predominantly occur in the southern parts of each of these BDMPS areas. Numbers in the NW part of the North Sea BDMPS are very small. About 100 pairs breed in this area, with 41 pairs on SPAs (Ythan Estuary, Firth of Tay). Numbers in the West of Scotland part of

the UK western waters BDMPS are also small. About 220 pairs breed in this area, with about 19 pairs on SPAs (Monach Isles, S Uist Machair).

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 60 and 61.

Based on evidence reviewed in sections 20.5, 20.6 and 20.7, the UK North Sea and Channel migration seasons BDMPS is estimated to hold 100% of adults and 60% of immatures from colonies in the UK North Sea and Channel, but no birds from colonies in the UK western waters area or from Ireland (Appendix A Table 60). These proportions give an estimated BDMPS of 3,524 birds (adults and immatures) with 3,524 of these from the UK population and none from overseas.

Based on evidence reviewed in sections 20.5, 20.6 and 20.7, the UK western waters migration seasons BDMPS is estimated to no birds from colonies in the UK North Sea and Channel, but 100% of adults and 60% of immatures from colonies in the UK western waters area, and 95% of adults and 60% of immatures from Ireland (Appendix A Table 61). These proportions give an estimated BDMPS of 1,602 birds (adults and immatures) with 1,088 of these from the UK population and 514 from overseas.

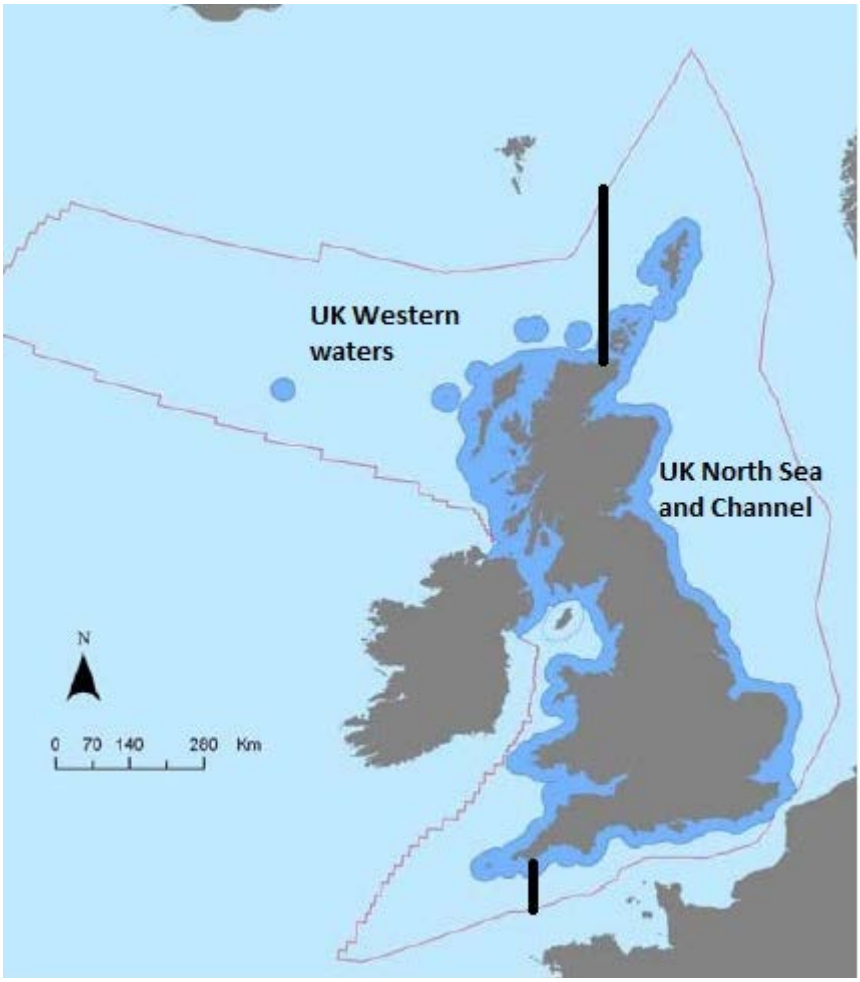


Figure 20.6. Two defined BDMPS spatial areas for little tern: ‘UK North Sea waters and Channel’ and ‘UK Western waters’.

20.11 Proportions of UK SPA birds in BDMPS

SPA birds represent about 40% of the UK population. Proportions of birds that are adults from UK SPA colonies in each BDMPS can be estimated directly from the data in Appendix A Tables 60 and 61. For example, in the UK North Sea and Channel BDMPS (3,524 birds) there are estimated to be 1,918 adults from SPA colonies, so these represent 54% of the total birds present.

20.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Although the general migration pattern of little terns is understood and colony sizes are rather well documented, almost nothing is known about the details of local (colony-specific) patterns of dispersal and migration behaviour of little terns through UK waters. However, it seems likely that birds will mix across the BDMPS when away from colonies, particularly because there are numerous but mostly fairly small colonies in each of the two BDMPS areas.

21. COMMON GUILLEMOT *Uria aalge*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (August to February) (adults and immatures)
Overseas	993,000	128,360
UK	3,132,000	2,628,166
Total	4,125,000	2,756,526

Non-breeding season BDMPS (August to February)	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
UK North Sea and Channel	1,617,306	94,160	1,523,146
UK Western waters	1,139,220	34,200	1,105,020

Colour coding is amber for numbers of birds in the UK population in the biogeographic total and in UK waters and each BDMPS since the locations and sizes of colonies in the UK are well known. Only a few colonies have not been censused since Seabird 2000, and population monitoring by JNCC has a strong focus on common guillemot so national and regional trends in numbers are well monitored. Dispersal and migratory movements of common guillemots from UK colonies are broadly well known based on ring recovery data, seawatching and at sea observations, although there is evidence for long term changes in migration patterns that relate to changes in availability of small pelagic fish (e.g. Heubeck et al. 1991), and the details of post-breeding dispersal of males with chicks are not well understood at a local level where interactions with renewables might be an issue as birds disperse rapidly from breeding areas. Numbers of birds from overseas populations that visit UK waters are much less well known, and there is much more uncertainty about population sizes in many overseas populations and whether those numbers are changing. Therefore the data for overseas contributions to the biogeographic population and BDMPS are coded red. However, because total numbers in the BDMPS are mainly determined by numbers in the UK component of the BDMPS, the totals are coded amber rather than red, as the influence of uncertainty in numbers from overseas on the total numbers present seems to be relatively small.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 62 and 63.

21.1 Breeding range and taxa

Common guillemot has a Holarctic breeding distribution. There are five subspecies, three of which occur in UK waters. *U. a. hyperborea* breeds in Svalbard and northern Norway, east to Novaya Zemlya in northern Russia, and has been recorded in very small numbers in the UK in winter. *U. a. albionis* breeds in Ireland, Britain south of 55° 38'N, at Helgoland (Germany) and from Brittany to northern Portugal. Nominate *U. a. aalge* breeds in Britain north of 55° 38'N, in southern Norway and the Baltic Sea (Peterz and Blomqvist 2010), Faroe, Iceland, Greenland and the northern Atlantic coast of North America. However, these subspecies may really represent clinal variation in size and plumage rather than discrete types, as colonies close to 55° 38'N may contain a mixture of birds that could be assigned to either *albionis* or *aalge*, and several chicks ringed in one subspecies have subsequently recruited into a colony of a different subspecies (for example *aalge* from Shetland found breeding in Arctic Norway where the subspecies is *hyperborea*). However, birds can generally be identified to subspecies from plumage and biometrics, and there is clinal variation in size (Hope Jones 1988, 1995) with larger birds further north, and in the presence of 'bridled' plumage with a higher frequency further north (Birkhead 1984; Reiertsen et al. 2012). As a result, there is scope to assess origins of birds sampled in winter (most frequently from beached birds associated with oil spills or winter wrecks or as a result of chronic winter mortality), although Barrett et al. (2008) concluded that biometrics only allow the most likely sea area of origin to be estimated rather than the specific colony. Attempts have also been made to use DNA markers to identify origins of common guillemots, but there is little variation in common guillemot DNA between populations (Moum et al. 1991; Moum and Arnason 2001; Cadiou et al. 2004; Riffaut et al. 2005).

21.2 Non-breeding component of the population

Common guillemots start to breed when 5 years old (BTO Birdfacts). Adult survival rate is 0.946 (BTO Birdfacts; Harris et al. 2000), juvenile survival 0.56 (BTO Birdfacts; Harris et al. 2007) and mean productivity is 0.678 chicks per pair (JNCC database, n=191 measurements). To obtain a stable population, survival of immatures was adjusted to 0.5 for juveniles, 0.6 for 1-year olds, 0.7 for 2-year olds, 0.85 for 3-year olds and 0.9 for 4-year olds. The model population comprised 57% adults, 19% juveniles and 24% older immatures. There are 0.74 immatures per adult.

21.3 Phenology

Breeding colonies in the UK are deserted in August, with modal departure in July (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007). Autumn dispersal/migration starts in mid-July (Cramp et al. 1977-94), July (Pennington et al. 2004), late July (Forrester et al. 2007) or August (Wernham et al. 2002). The late start date noted by Wernham et al. (2002) may be because that analysis is based primarily on ring recoveries, and there may be a lag before recoveries are found. Peak autumn migration occurs in August according to Pennington et al. (2004) and Forrester et al. (2007), in August-October (Cramp et al. 1977-94), in September-October (Wernham et al. 2002), or October-December in Belgium (Vanermen et al. 2013). Numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) are remarkably small, and do not indicate timing of autumn migration, presumably because birds from breeding sites move eastwards across the North Sea rather than southwards along the coast past most of these seawatching sites (Figure 21.1). Autumn migration is completed by September (Pennington et al. 2004; Forrester et al. 2007) or October (Cramp et al. 1977-94) or November (Wernham et al. 2002). Again the estimate from ring recovery data may be a little late by comparison with estimates based on direct observations.

Spring migration starts in October-November (Cramp et al. 1977-94; Pennington et al. 2004), October-February (Forrester et al. 2007) or December (Wernham et al. 2002). Peak spring migration occurs in December-February (Cramp et al. 1977-94; Forrester et al. 2007),

January-February (Pennington et al. 2004), January-March (Wernham et al. 2002) and January-March in Belgium (Vanermen et al. 2013). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in early February (Figure 21.1). Spring migration is completed by March (Pennington et al. 2004; Forrester et al. 2007), mid-April (Cramp et al. 1977-94), or April-May (Wernham et al. 2002).

The first spring records of common guillemot in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were predominantly from 1 January and the last records were predominantly at 31 December, as large numbers of common guillemots overwinter, while peak autumn migration was reported in July in most years, and peak spring migration was reported in January-March in most years. Birds re-occupy colonies from November, with modal return in January (Mudge et al. 1987; Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007).

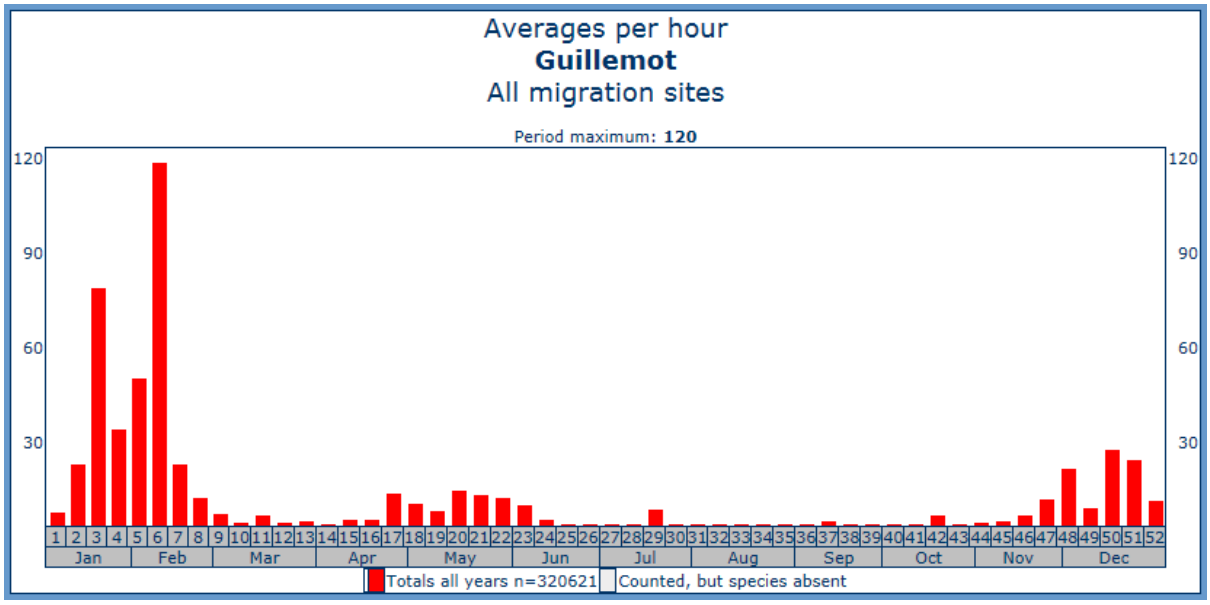


Figure 21.1. Average numbers of common guillemots counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-June, non-breeding season October-April. However, from the data reviewed above, this could be refined to breeding season March-July, non-breeding season August-February.

21.4 Defined seasons:

- UK Breeding season — March-July
- Post-breeding migration in UK waters — July-October
- non-breeding season — August-February (**non-breeding BDMPS**)
- Return migration through UK waters — December-February
- Migration-free breeding season — March-June
- Migration-free winter season — November

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for common guillemot:

Non-breeding season BDMPS (August-February).

21.5 Movements of birds from the UK population

Common guillemots in Britain and Ireland are considered to be dispersive rather than migratory (Wernham et al. 2002). Many adults remain close to their colony throughout the year (Brown and Grice 2005). With the exception of August-September, when adults moult and are flightless for about six to seven weeks (Brown and Grice 2005), adults can be seen at breeding sites occupying ledges, though sporadically through winter. Young birds disperse further than adults, and juveniles from UK colonies have been recovered in October onwards from north Norway to Portugal, whereas few adults move beyond UK waters (Wernham et al. 2002). There is a slight indication for birds from different parts of the UK wintering in different areas, as suggested by Mead (1974). Birds from northern Britain move furthest (and include most of the recoveries in north Norway) (Wernham et al. 2002; see also Heubeck et al. 1991). Those from colonies in SW England mostly move southwards into the Bay of Biscay and travel least (Wernham et al. 2002). Common guillemots from colonies in the east coasts of England and Scotland mostly remain in the North Sea in winter (Wernham et al. 2002). Although typical patterns of distribution and seasonal movements are described above, there is very strong evidence indicating that common guillemot seasonal movements, distribution patterns and overwinter survival are strongly affected by the distribution and abundance of prey fish stocks, and especially the distribution and abundance of sprats. Since sprat stock biomass can vary considerably from year to year, common guillemot seasonal movements can vary according to the availability of their winter prey. Blake (1984) suggested that guillemot survival in winter was influenced by abundance of small prey fish stocks within local areas. Mass mortality of guillemots in 1983 correlated with apparent low abundance of sprat, one of their main winter foods in areas of the North Sea (Underwood and Stowe 1984). Blake et al. (1984) suggested that guillemot distribution across the North Sea related to presence of sprat stocks, while Peterz and Olden (1987) found that increased numbers of common guillemots wintering off the west coast of Sweden related to high abundance of young herring in that area at the time. Skov et al. (2000) also found that the distribution of common guillemots in winter in the Skagerrak and Kattegat correlated with the distribution of young herring. Harris and Bailey (1992) showed that first year common guillemot survival rates in the North Sea were best explained by sprat stock biomass. Although Pennington et al. (2004) stated that the breeding numbers and breeding success of common guillemots in Shetland was primarily determined by the biomass of the Shetland sandeel stock, sandeels remain buried in the sea bed during autumn and winter so are not readily available at that time of year (although common guillemots have been recorded to dig sandeels out of the sand in winter). Their winter prey is predominantly sprats and young herring (Blake 1984).

When common guillemot chicks fledge from Shetland colonies in July, in most years the chicks swim eastwards accompanied by the male parent, arriving off the coast of Norway within a few weeks (Pennington 2004). During 1982-84, many thousands remained in inshore waters around Shetland instead of travelling to Norway. This altered behaviour coincided with a high abundance of sandeels at Shetland and low sprat biomass in the North Sea. No such large numbers were encountered there post-fledging during the late 1980s or 1990s when sandeel stocks had declined to very low abundance at Shetland. These observations suggest that the movements and resulting winter distribution of common guillemots, perhaps especially first year birds, are highly flexible, with birds aggregating in areas where there are high concentrations of food fish. In English waters, post-breeding aggregations are particularly found in August over Dogger Bank, off East England northwards of Flamborough, and in the Irish Sea (Brown and Grice 2005); these birds become more widely dispersed from October to February. There is concern that common guillemots dispersing from breeding areas may possibly aggregate in, or pass through, sites being considered for marine renewables development during their dispersal phase. Since that can be very rapid, lasting just two or three weeks in July, such aggregations could easily be overlooked by a survey protocol of monthly counts at a proposed development site, while

such aggregations may not necessarily occur in the same place in successive years, depending on fish stocks. There is, therefore, much uncertainty about local aggregations post-breeding, and where these might be located. More work is required to map dispersal by males and chicks before it is possible to define a BDMPS or set of BDMPSs for the dispersal phase. Therefore in this report the dispersal phase is not treated separately, but is subsumed into the defined breeding season (March to July) or non-breeding season (August to February).

21.6 Movements of birds from overseas into UK waters

Wernham et al. (2002) report 69 foreign ringed common guillemots recovered in the British Isles (i.e. not including birds ringed in Ireland as foreign). These included 8 ringed in Netherlands, 29 in Germany, 23 in Faroes, 5 in Norway, 3 in France, and 1 in Russia. Some of these were ringed as rehabilitated birds (e.g. those from Netherlands, and probably those from France and some from Germany). From this they concluded that small numbers of common guillemots from Scandinavian and Faroese colonies reach northern Britain in autumn and winter and some enter the North Sea (Wernham et al. 2002). Deployment of geolocators on breeding common guillemots at colonies in Central Norway found that all moved northwards up the Norwegian Sea after the breeding season (Lorentsen and May 2012). 80% of these then moved into the Barents Sea, while 20% remained in the north Norwegian Sea. After moult, some moved back into the north Norwegian Sea so that 50% overwintered in the Barents Sea and 50% in the north Norwegian Sea. Lorentsen and May (2012) point out that there are ring recoveries of common guillemots from Central Norway in southern Norway as well as to the north, and caution that their geolocator results may represent only the year of deployment (2009-10) and that patterns may differ in other years, but they suggest that the Barents Sea may represent the main moulting area and a major wintering area for common guillemots from colonies in Central Norway (see also Steen et al. 2013). Most of the recoveries abroad of common guillemots ringed in the Faroes that were recovered in September to November were from the coast of Norway, with only two from UK coasts (Hammer et al. 2013). Later in the winter, in December to February, 7 were recovered on UK North Sea coasts (including Shetland), 18 on the Norwegian coast, 1 in Denmark and 1 in Iceland (Hammer et al. 2013). Birds from Germany (Helgoland) winter in the North Sea and some may enter UK waters (Wernham et al. 2002). The Baltic population apparently remains within the Baltic Sea (Wernham et al. 2002). Wernham et al. (2002) did not report any common guillemots from Iceland recovered in UK waters. Pennington et al. (2004) reported that the only foreign-ringed common guillemots recovered in Shetland were three birds ringed in Faroe. A very few birds found in Shetland in winter appear from measurements to be from the subspecies *hyperborea* which breeds in Arctic Norway, Bear Island, Svalbard and northern Russia (Pennington et al. 2004) but these have only been found on a very few occasions so numbers coming from far northern populations appear to be negligible. Fort et al. (2013) report on deployment of geolocators on common guillemots breeding at a northern Barents Sea colony. Those birds remained within the Barents Sea, White Sea or north Norwegian Sea throughout the winter, so geolocator data suggest that high latitude common guillemots are unlikely to reach UK waters except as vagrants. This supports conclusions based on ringing, which also indicated that common guillemots from colonies in the southern Barents Sea (north Norwegian coast) spend the winter either in the Barents Sea, or in the north Norwegian Sea (Nikolaeva et al. 1996). A small number of birds ringed as chicks have been recovered at breeding colonies far from their natal origins; two chicks from UK colonies were recovered at a colony in north Norway, two from UK colonies were recovered at colonies in the Baltic, and one from the Baltic bred at Skomer in the Irish Sea (Wernham et al. 2002). However, these long-distance natal dispersals are very exceptional. Anker-Nilssen et al. (1988) used biometrics of 826 common guillemots (18% of which were adults) killed by oil in the Skagerrak in January 1981 to infer that most were probably from Scottish or south Norwegian colonies. Cadiou et al. (2004) used ring recoveries and biometrics of 1,851 common guillemots killed in the 'Erika' oil spill in the Bay

of Biscay to infer that birds originated from a large area that included colonies from across the British Isles, along with some from more northerly colonies, but with most birds coming from colonies between west Scotland and the Celtic Sea. Grantham (2004) identified differences in wintering areas used by birds from different colonies as recovered in major oil spills; birds wintering in the southwestern approaches to the English Channel and in the Bay of Biscay tended to be immature birds from colonies in west Britain and Ireland, whereas birds wintering in the English Channel and southern North Sea tended to be adults from colonies in eastern Britain. Seabird 2000 reported 965,000 pairs in UK, 80,000 pairs in Ireland, 101,000 pairs in Norway, 175,000 pairs in Faroe, 990,000 pairs in Iceland, 2,500 pairs in Germany, 2,500 pairs in Denmark, and 250 pairs in France (Mitchell et al. 2004). More recently, Hammer et al. (2013) estimated that there are about 100,000 pairs in Faroe, while Gardarsson (2006) suggested that breeding numbers in Iceland had declined by 30% between 1983-86 and 2005-08, with 693,000 pairs in 2005-08. In the UK, changes in numbers are uncertain as no complete survey has been carried out since 2000, but JNCC monitoring data from a selection of colonies suggest a decrease in breeding numbers of about 40% in Scotland between 2000 and 2011 with the decrease most evident in Shetland (Foster and Marrs 2012) whereas numbers breeding in Wales have increased by a similar percentage (JNCC database).

21.7 Numbers in UK waters

Forrester et al. (2007) suggest that in winter there are around 750,000 individuals of *Uria aalge aalge* in Scottish waters. Numbers of *Uria aalge albionis* in Scottish waters in winter are uncertain, but there may be around 20,000, with most of those birds (which breed mostly on Ailsa Craig and Sanda) being in SW Scotland in winter. A small proportion of those birds may originate from colonies in England, Wales and Ireland, as some of those birds may disperse northwards in autumn (Forrester et al. 2007). Blake et al. (1984) estimated from ESAS data that common guillemots moved rapidly out of waters adjacent to breeding colonies in July, with perhaps 1,500,000 birds in North Sea waters in autumn and winter. Numbers in waters to the west of the UK appear to be similar in total to numbers in UK North Sea waters, so perhaps about 1,500,000 birds are in waters west of the UK in autumn and winter. Those totals would suggest that most of the UK population (900,000 pairs so 1,800,000 adults which would probably have an associated 1,300,000 immature birds) are in UK waters in autumn and winter, or that the birds from the UK population that move into overseas waters are similar in number to the totals that enter UK waters from overseas. This total would suggest that the estimate presented by Forrester et al. (2007) is most likely an underestimate of numbers in Scottish waters. For this reason, estimated numbers in the BDMPS have been set between the (lower) numbers thought to be at sea based on ESAS survey data and (higher) numbers thought to be present based on known population size and movement patterns.

21.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the North Atlantic population, comprising 2,250,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 2,800,000-2,900,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 8,500,000 individuals. Populations with connectivity to UK waters include UK (900,000 pairs), Ireland (80,000 pairs), Faroe (100,000 pairs), Norway (100,000 pairs), Germany and Denmark (5,000 pairs) and France (250 pairs). Therefore the biogeographic population with connectivity to UK waters sums to 4,125,000 birds (including adults and immatures), with 3,132,000 in UK, and 993,000 in overseas, populations. The UK population represents a high proportion of this total, and many of the birds from these overseas populations do not visit UK waters, so the birds in UK waters are, at all times of year, predominantly birds from UK colonies. The estimated total numbers in UK waters in the non-breeding season (August to February) are 2,708,000 birds, with 2,580,000 of these from the UK. The slightly smaller number of UK birds in UK waters than

in the biogeographic population recognises that some younger immature birds from the UK will be in overseas waters.



Figure 21.2. Breeding population origins of common guillemots in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

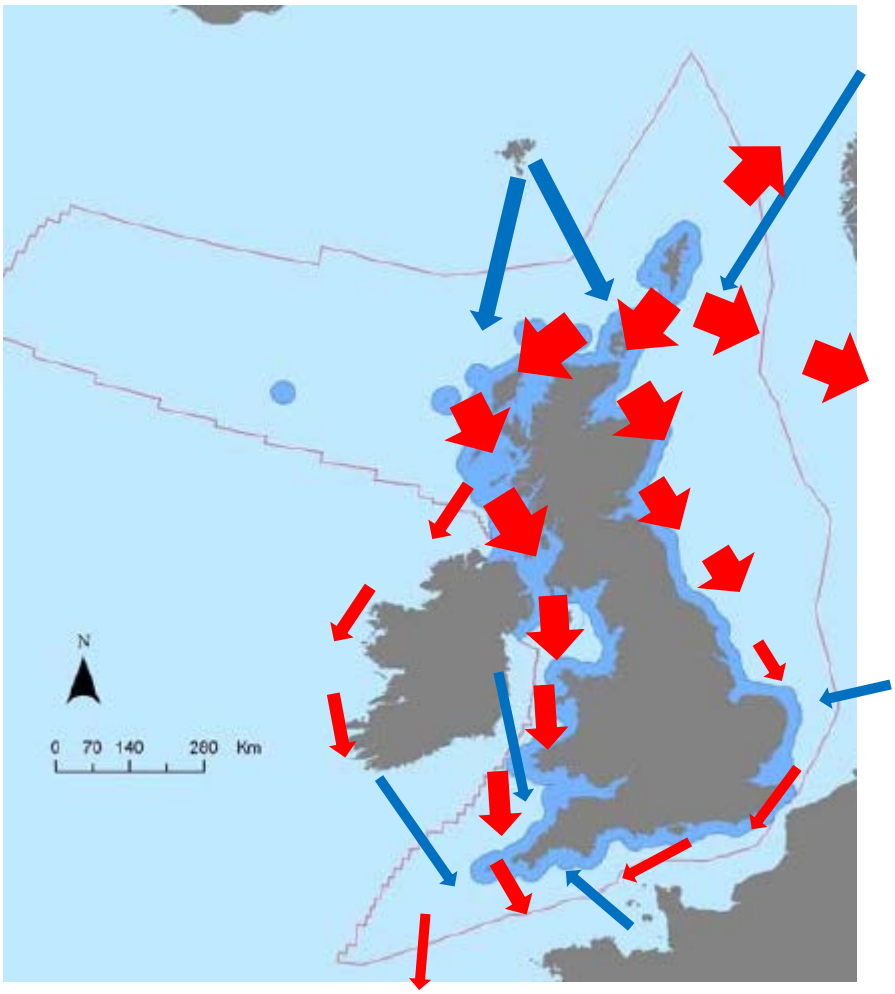


Figure 21.3. Main movements of common guillemots from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

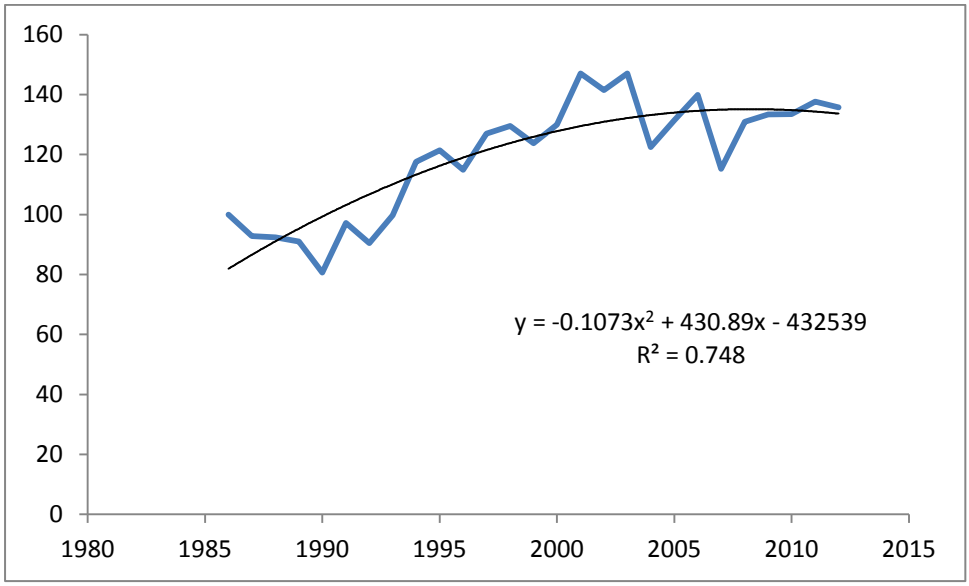


Figure 21.4. Trend in the common guillemot breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

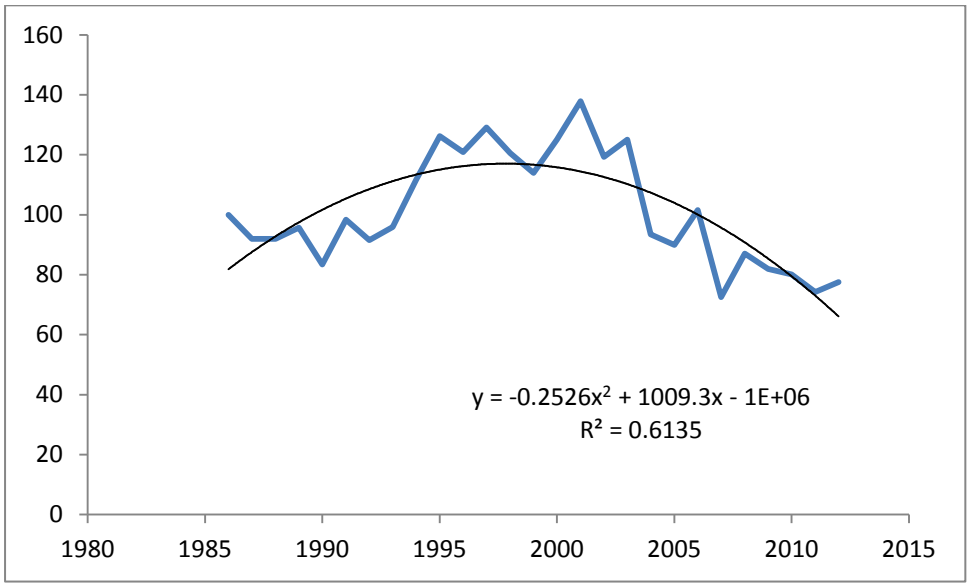


Figure 21.5. Trend in the common guillemot breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

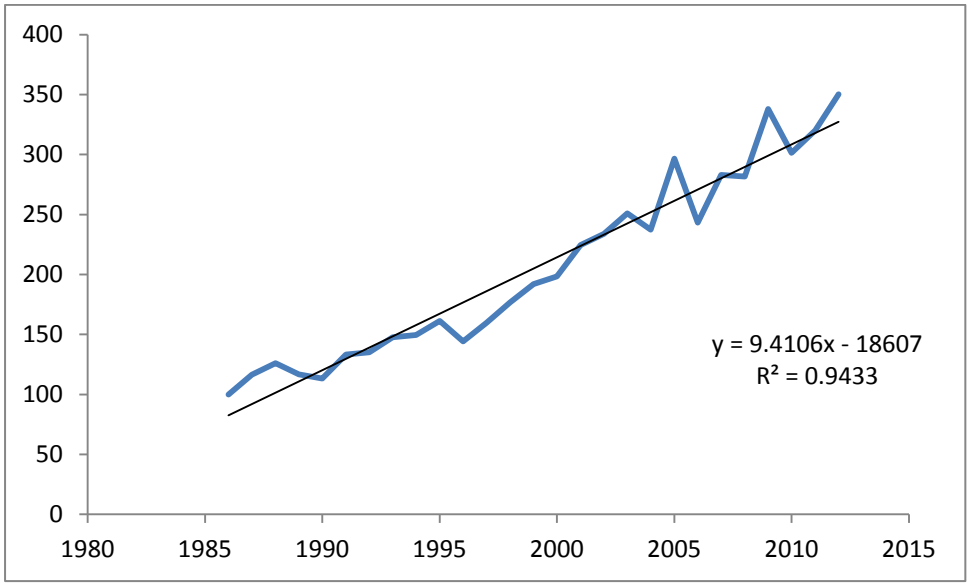


Figure 21.6. Trend in the common guillemot breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

21.9 Proportion of BDMPS from UK breeding SPAs

The 34 SPAs with breeding common guillemots as a feature together held 693,120 pairs at designation, estimated to represent ca. 95% of the British breeding population and ca. 27% of the all-Ireland breeding population (Stroud et al. 2001). Stroud et al. (2014) considered the two subspecies that occur in the UK separately. The subspecies *Uria aalge aalge* breeding populations are features in 30 GB SPAs, and survey data from 1999-2011 showed that those then held an estimated 75% of the GB population of that subspecies. The subspecies *Uria aalge albionis* breeding populations are features in 3 GB SPAs, and survey data from 2009-2011 showed that those then held an estimated 68% of the GB population of that subspecies. The single SPA for *Uria aalge albionis* in Northern Ireland then held an estimated 55% of the all-Ireland population of that subspecies. Since the surveys reported in Stroud et al. (2014) numbers have declined further in northern Scotland but increased in England and Wales; the proportion in the SPA suite may have further reduced slightly, but probably very little overall.

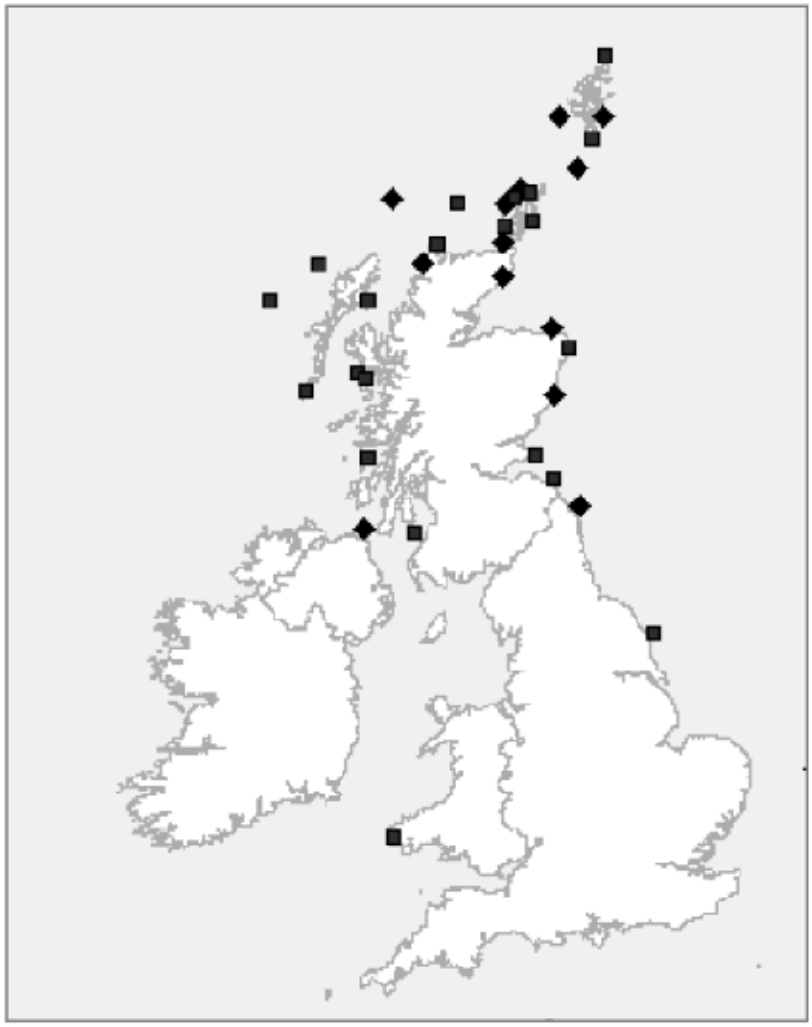


Figure 21.7. The UK SPA suite for breeding common guillemots. These SPA populations are listed in Table 21.1.

Table 21.1. The UK SPA suite for breeding common guillemots (counts expressed as individual birds are converted to pairs by multiplying by 0.67).

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count (pairs)	Year	Reference
UK North Sea & Channel							
Hermaness, Saxavord & Valla	Shetland	11,363	1994	Maintained 2000	6,994 4,020 4,620	2000 2004 2009	SMP database SMP database SMP database
Foula	Shetland	25,125 (1987)	1995	Declined 2007	27,805 16,615	2000 2007	SMP database SMP database
Noss	Shetland	30,619	1996	Declined 2005	30,671 14,908 16,172 14,783	2001 2004 2005 2009	SMP database SMP database SMP database SMP database
Sumburgh Head	Shetland	10,752	1996	Declined 2007	10,269 5,109 4,908 5,314 4,762 3,323 4,896 4,207	2001 2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database

Fair Isle	Shetland	25,165 (1994)	1994	Maintained 1999	26,302 18,304 13,066	1999 2005 2010	SMP database SMP database SMP database
West Westray	Orkney	28,274	1996	Maintained 2007	36,700 33,900	1999 2007	Seabird2000 Lewis et al. 2012
Calf of Eday	Orkney	8,241	1998	No change 2006	1,715 6,300	2002 2006	SMP database Lewis et al. 2012
Rousay	Orkney	7,102	2000	Recovered 2009	4,300 6,200	1999 2009	Seabird2000 Lewis et al. 2012
Marwick Head	Orkney	24,388	1994	Maintained 1999	23,235 7,019 11,267 11,097	1999 2004 2006 2012	SMP database SMP database SMP database SMP database
Hoy	Orkney	13,400	2000	Declined 2007	6,300	2007	Lewis et al. 2012
Copinsay	Orkney	13,333	1994	Declined 2008	9,166 5,607	2008 2012	SMP database SMP database
North Caithness Cliffs	N Scotland	26,994	1996	Maintained 2000	47,000	2000	Lewis et al. 2012
East Caithness Cliffs	N Scotland	71,509 (1986)	1996	Maintained 1999	120,789 or 158,895 individuals	1999	Lewis et al. 2012
Troup, Pennan & Lion's Heads	NE Scotland	29,902 (1995)	1997	Declined 2007	30,300 10,938	2001 2007	Seabird2000 SMP database
Buchan Ness - Collieston Coast	NE Scotland	8,640	1998	Declining 2007	19,691 12,928	2001 2007	SMP database SMP database
Fowlsheugh	NE Scotland	40,140	1992	Maintained 1999	41,800 36,300 33,900 30,100	1999 2006 2009 2012	SMP database SMP database SMP database SMP database
Forth Islands	E Scotland	16,000 (1985) Or 22,452 (Stroud et al. 2001)	1990	Maintained 2007	14,096 15,829 16,091 15,779 14,674	2007 2008 2009 2010 2011	SMP database SMP database SMP database SMP database SMP database
St Abb's Head to Fast Castle	E Scotland	20,971	1997	Maintained 1998	27,282 27,061 22,231 22,103	1998 2003 2008 2013	SMP database SMP database SMP database SMP database
Farne Islands	NE England	23,499	1985		32,596 29,390 32,244 31,058 32,145 32,881 33,532	2007 2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database

Flamborough Head & Bempton (to be subsumed into Flamborough and Filey Coast SPA)	E England	16,150	1993		31,279 39,641	2000 2008	SMP database SMP database
Flamborough and Filey Coast	E England	41,607 (2008-2011)	Not yet				See row above
UK Western waters							
Sule Skerry and Sule Stack	N Scotland	6,298 (1986)	1994	Maintained 1998	7,633	1998	SMP database
North Rona and Sula Sgeir	N Scotland	28,944 (1986)	2001	Declined 2012	21,021 North Rona only: 7,033 4,096 3,324	1998 1998 2005 2012	SMP database SMP database SMP database SMP database
Cape Wrath	NW Scotland	9,159	1996	Maintained 2000	27,359	2000	SMP database
Handa	NW Scotland	76,105 (1994)	1990	Declined 2007	75,493 60,370 30,550 37,993	1998 2003 2007 2011	SMP database SMP database SMP database SMP database
Shiant Isles	Western Isles	12,315	1992	Declined 2008	11,026 5,148	1999 2008	SMP database SMP database
Flannan Isles	Western Isles	14,693	1992	Declined 2013	9,807	1998	Mitchell et al. 2004
St Kilda	Western Isles	15,209	1992	Maintained 2000	15,700	1999	Seabird2000
Canna and Sanday	W Scotland	3,858	1998	Maintained 2001	3,913	1999	SMP database
Rum	W Scotland	2,680	1982	No change 2000	1,644	2000	SMP database
Mingulay and Berneray	Western Isles	20,703	1994	Declined 2009	21,835 29,725 13,527	1998 2003 2009	SMP database SMP database SMP database
North Colonsay & West Cliffs	W Scotland	6,656	1997	Maintained 2008	13,500	2000	Seabird2000
Ailsa Craig	W Scotland	3,350 (1987)	1990	Maintained 2003	7,818 5,247	2009 2013	SMP database SMP database
Rathlin Island	N Ireland	28,064 (1985)	1999		54,473 87,398	2007 2011	SMP database SMP database
Skomer and Skokholm	Wales	7,067	1982		12,479 14,210 14,577 16,375 16,641 16,300	2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

21.10 BDMPS

UK waters can be split into two spatial BDMPS for common guillemots, the UK North Sea and Channel, and the UK western waters (Figure 21.8). This split is based on the fact that very few common guillemots from colonies in western Britain move into the North Sea during autumn migration or vice versa. In addition, birds from overseas are likely to show a tendency to occur more in one side of the UK than the other, with birds from continental Europe more frequent in the North Sea than in western waters. While there is a possibility that spatial distribution patterns may differ in the immediate post-breeding dispersal period in July-August, the details of distribution and movements at that time are not well known except broadly. There have not yet been any tracking studies of males with dependent chicks as they disperse, so details of colony-specific patterns of dispersal and how much these vary from year to year are uncertain. Until such data are available it seems best to define just two seasonal periods; breeding season (March to July) and non-breeding season (August to February).

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 62 and 63.

Based on evidence reviewed in sections 21.5, 21.6 and 21.7, the UK North Sea and Channel non-breeding season BDMPS is estimated to hold 70% of adults and 60% of immatures from North Sea colonies in Shetland to Aberdeenshire, 80% of adults and 70% of immatures from Aberdeenshire to Fife, 90% of adults and 80% of immatures from Fife to Humberside, 5% of adults and 10% of immatures from colonies from NW Scotland to Argyll, 0% of adults and 5% of immatures from Argyll to Northern Ireland, 5% of adults and 10% of immatures from Wales, 10% of adults and 20% of immatures from Faroes, 5% of adults and 20% of immatures from Norway, 20% of adults and 40% of immatures from Germany and Denmark (Appendix A Table 62). These proportions result in an estimated BDMPS of 1,617,306 birds (adults and immatures) with 1,523,146 of these from UK and 94,160 from overseas populations.

Based on evidence reviewed in sections 21.5, 21.6 and 21.7, the UK western waters non-breeding season BDMPS is estimated to hold 2% of adults and 5% of immatures from North Sea colonies in Shetland, Orkney and north Caithness, no birds from colonies between East Caithness and East Anglia, 95% of adults and 90% of immatures from colonies from NW Scotland to Argyll, 100% of adults and 95% of immatures from Argyll to Northern Ireland, 90% of adults and 80% of immatures from Wales, 5% of adults and 10% of immatures from Faroes, 1% of adults and 5% of immatures from Norway (Appendix A Table 63). These proportions result in an estimated BDMPS of 1,139,220 birds (adults and immatures) with 1,105,020 of these from UK and 34,200 from overseas populations.

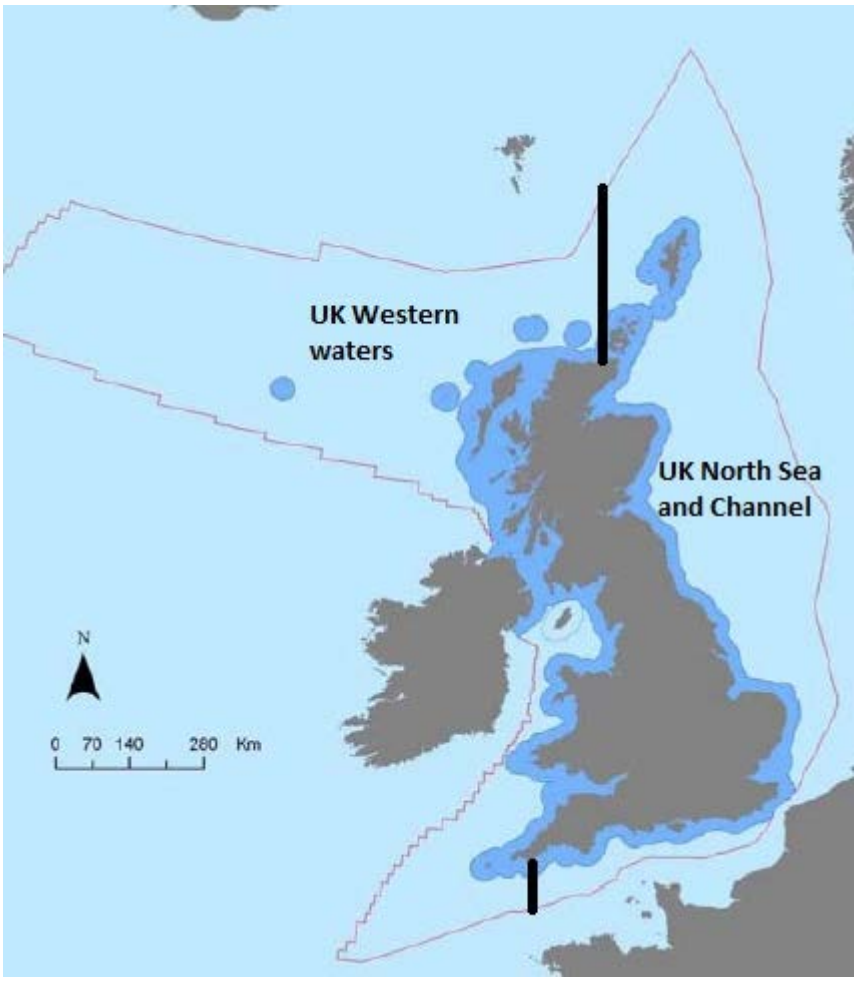


Figure 21.8. Two defined BDMPS spatial areas for common guillemot: 'UK North Sea waters and Channel' and 'UK Western waters'.

21.11 Proportions of UK SPA birds in BDMPS

SPA birds represent about 70-75% of the UK population. Proportions of birds that are adults from UK SPA colonies in each BDMPS can be estimated directly from the data in Appendix A Tables 62 and 63. For example, in the UK North Sea and Channel BDMPS (1,617,306 birds) there are estimated to be 684,920 adults from SPA colonies, so these represent 42% of the total birds present.

21.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Given the large number of SPA populations distributed through UK coasts, the SPA birds are likely to be well mixed with birds from non-SPA colonies and from overseas. In autumn shortly after dispersal from colonies there may be aggregations of SPA birds close to Flamborough Head & Bempton SPA, close to Farne Islands SPA, and close to Skokholm and Skomer SPA. These aggregations are likely to become less pronounced through the autumn as birds move offshore during winter, but may recur in late winter as adult birds move back towards breeding colonies. However, such aggregations appear to be very short-lived in the transition between breeding and non-breeding distributions. More research is needed to determine whether there are consistent 'hot-spots' where common guillemots aggregate during the brief post-breeding dispersal stage in late July.

22. RAZORBILL *Alca torda*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in migration seasons (August-October, and January-March) (adults and immatures)	Numbers in UK waters in winter (November-December) (adults and immatures)
Overseas	1,350,000	851,310	461,228
UK	357,000	347,478	98,816
Total	1,707,000	1,198,788	560,044

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Migration seasons BDMPS (August-October, and January-March)			
UK North Sea and Channel	591,874	434,431	157,443
UK Western waters	606,914	416,879	190,035
Winter BDMPS (November and December)			
UK North Sea and Channel	218,622	172,869	45,753
UK Western waters	341,422	288,359	53,063

Colour coding is amber for numbers of birds in the UK population in the biogeographic total and in UK waters and each BDMPS since the locations and sizes of colonies in the UK are well known. Only a few colonies have not been censused since Seabird 2000, and population monitoring by JNCC has meant that national and regional trends in numbers can be assessed. Dispersal and migratory movements of razorbills from UK colonies are broadly known based on ring recovery data, seawatching and at sea observations, although there may be long term changes in migration patterns that relate to changes in availability of small pelagic fish (as is more clearly known for common guillemot), and the details of post-breeding dispersal of males with chicks are not well understood at a local level where interactions with renewables might be an issue as birds disperse rapidly from breeding areas. Numbers of birds from overseas populations that visit UK waters are much less well known, and there is much more uncertainty about population sizes in many overseas populations and whether those numbers are changing. Therefore the data for overseas contributions to the biogeographic population and BDMPS are coded red. Because it

appears that total numbers are strongly influenced by these numbers from overseas, the totals are also coded red.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 64 to 67.

22.1 Breeding range and taxa

Razorbills breed around the North Atlantic. There are two subspecies; nominate *A. t. torda* breeds in eastern North America, Greenland, Bear Island, White Sea, Norway, Denmark, and Baltic Sea. Subspecies *islandica* breeds in in Iceland, Faroe, British Isles, Germany, and France. There is considerable variation in size with latitude of breeding colony (Hope Jones 1995; Barrett et al. 1997), providing an opportunity to assess origins of individuals sampled in winter. Although genetic differentiation between razorbills in different colonies was considered by Moum and Arnason (2001) to be moderately high, genetic comparisons do not seem to have been used to infer seasonal movements of razorbills.

22.2 Non-breeding component of the population

Razorbills start to breed when 4 years old (BTO Birdfacts). Adult survival rate is 0.9 (BTO Birdfacts; Chapdelaine 1997), juvenile survival 0.38 to 4 years old (BTO Birdfacts; Chapdelaine 1997) and mean productivity is 0.633 chicks per pair (JNCC database, n=87 measurements). To obtain a stable population, survival of immatures was adjusted to 0.6 for juveniles, 0.7 for 1-year olds, 0.8 for 2-year olds, and 0.9 for 3-year olds. The model population comprised 57% adults, 18% juveniles and 25% older immatures. There are 0.75 immatures per adult.

22.3 Phenology

Breeding colonies in the UK are deserted in August, with modal departure in July (Pennington et al. 2004; Forrester et al. 2007). Autumn dispersal/migration starts in July (Pennington et al. 2004), mid-July (Cramp et al. 1977-94), July-August (Wernham et al. 2002), or August (Forrester et al. 2007). Peak autumn migration occurs in late July in Shetland (Pennington et al. 2004), August-October (Cramp et al. 1977-94), September-October (Wernham et al. 2002; Forrester et al. 2007), and October-November in Belgium (Vanermen et al. 2013). Numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) are so small that no peak in autumn migration can be detected (Figure 22.1). Autumn migration is completed by mid-August in Shetland (Pennington et al. 2004), but by October in southern UK waters (Cramp et al. 1977-94), November (Forrester et al. 2007) or November-December (Wernham et al. 2002).

Spring migration starts in November-December (Cramp et al. 1977-94), or January (Pennington et al. 2004; Forrester et al. 2007). Peak spring migration occurs in January-February (Cramp et al. 1977-94), February-March (Pennington et al. 2004; Forrester et al. 2007), and February-April in Belgium (Vanermen et al. 2013). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in late January and early February (Figure 22.1). Spring migration is completed by March (Cramp et al. 1977-94) or April (Pennington et al. 2004; Forrester et al. 2007).

The first spring records of razorbill in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from January to March, and the last records were mostly in December. Peak autumn migration was not well defined and was reported in July to November in different areas and years, and peak spring migration was reported in January to April, but mostly in March. Birds re-occupy colonies from February, with modal return in late March or early April (Pennington et al. 2004; Forrester et al. 2007).

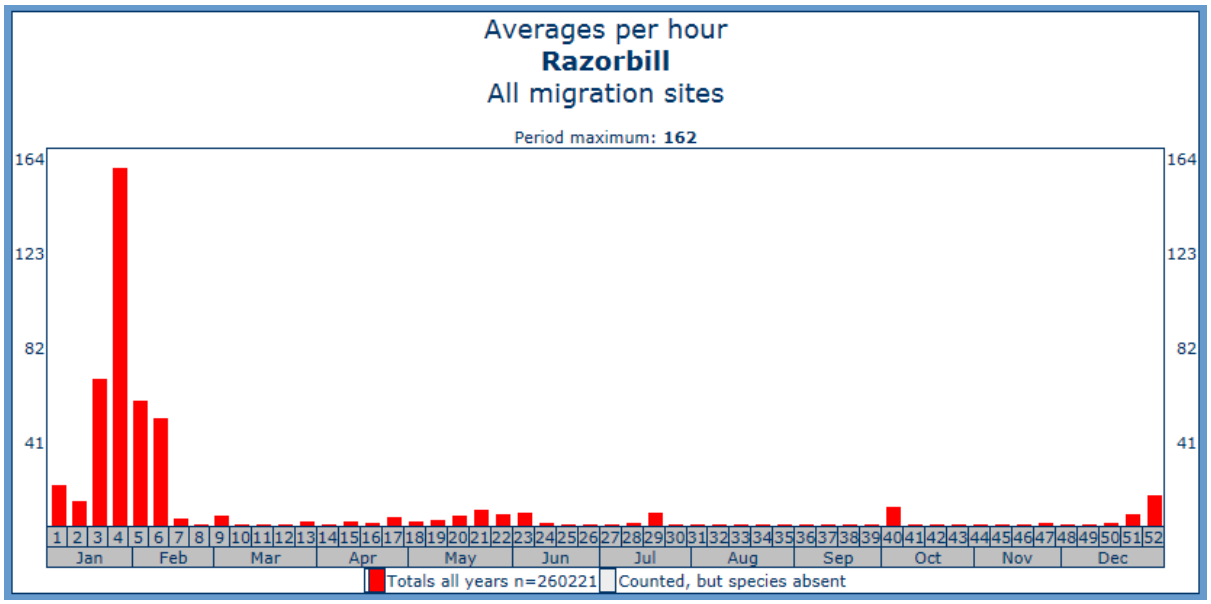


Figure 22.1. Average numbers of razorbills counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as May-June, non-breeding season October-April. However, from the data reviewed above, a more appropriate definition would be breeding season April-July, non-breeding season August-March.

22.4 Defined seasons:

- UK Breeding season — April-July
- Post-breeding migration in UK waters — August-October (**migration BDMPS**)
- non-breeding season — August-March
- Return migration through UK waters — January-March (**migration BDMPS**)
- Migration-free breeding season — April-June
- Migration-free winter season — November-December (**winter BDMPS**)

Apart from the breeding season, two seasonal BDMPS periods are considered to be appropriate for razorbill:

Migration seasons BDMPS (August-October, and January-March); and

Winter BDMPS (November-December).

22.5 Movements of birds from the UK population

During late summer and early autumn (July and August) when the fledged young are completing growth at sea and adults are undertaking their post-breeding moult, most recoveries of UK ringed adults and juveniles occur close to the colony, though by this time immature birds may be further afield (Wernham et al. 2002). During September, breeders and juveniles move predominantly southwards, with recoveries from southern Norway to Portugal, and predominantly in the southern North Sea, Celtic Sea, Channel or Bay of Biscay (Wernham et al. 2002). The majority of those ringed in the SW of Britain are recovered in autumn in the Channel, the southern North Sea, western France, Iberia, the western Mediterranean and northwest Africa. Razorbills from colonies in NW Britain are predominantly recovered from the North Sea, Channel, southern and western Britain and France. Birds from north Scotland and the northern isles tend to move east, to southwest

Norway and Denmark or to the southern North Sea with relatively few reaching France and Iberia. Skov et al. (2000) found that the distribution of razorbills in winter in the Skagerrak and Kattegat, some of which originate from UK colonies, correlated with the distribution of young herring. Too few birds have been ringed in east Britain to indicate their movement pattern. Immature birds, especially the youngest age classes, tend to travel further south in winter than adults, and may remain in wintering areas through the year, but older immatures tend to move back to breeding colonies in summer though some may visit areas beyond their natal colony such as Greenland, Iceland and Faroe. Adults return to their colonies in spring, with older immatures following later.

22.6 Movements of birds from overseas into UK waters

Only 26 razorbills ringed abroad have been recovered in Britain and Ireland; 14 of these were ringed in Iceland, 4 in Russia, 3 in Norway, 2 in France, and one each in Finland, Sweden and The Netherlands (Wernham et al. 2002). The birds from Russia, Norway, Finland and Sweden are from the subspecies *torda*, and these birds tend to be significantly larger than birds from the subspecies *islandica* which is found breeding in the UK, Iceland, Faroe, Ireland and France. Measurements of beached corpses of razorbills in winter have confirmed presence of birds of the subspecies *torda* at a frequency of up to 4% of beached razorbills in the British Isles in winter, suggesting that these larger birds from the nominate subspecies are present in UK waters in winter as a small minority of the razorbill population (Wernham et al. 2002). In Shetland, only very small numbers of birds with wing lengths indicative of the subspecies *torda* have been found in winter beached bird surveys, suggesting that rather few *torda* birds winter near to Shetland (Pennington et al. 2002). Anker-Nilssen et al. (1988) used biometrics of 308 razorbills (66% of which were adults) killed by oil in the Skagerrak in January 1981 to infer that 55% were probably from Scottish colonies, and 45% from Baltic colonies. Seabird 2000 reported populations as 126,400 pairs in UK, 17,000 pairs in Ireland, 380,000 pairs in Iceland (Gardarsson 2006 suggested this had decreased to 315,400 pairs by 2005-08), 4,500 pairs in Faroe (all these being populations of the subspecies *islandica*), 30,300 pairs in Norway, 3,500pairs in Russia, 10,000 pairs in Sweden, 6,000 pairs in Finland (all those being populations of the subspecies *torda*).

22.7 Numbers in UK waters

During post-breeding dispersal, about 220,000 birds are present in the North Sea (Tasker et al. 1987). Forrester et al. (2007) suggest that about 50,000 to 250,000 birds winter in Scottish waters, the high range indicating a low confidence in numbers. Higher numbers occur in English waters in winter, but a substantial (but uncertain) proportion of the UK population winters in southern Europe or in the eastern North Sea.

22.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the subspecies *islandica* population, comprising 575,000 pairs. However, Mitchell et al. (2004) provided a revised estimate of this population as 530,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 1,950,000 individuals. Populations with connectivity to UK waters in migration or winter (Figure 22.2) include the UK (120,000 pairs), Iceland (315,000 pairs), Faroe (4,500 pairs), Norway (30,300 pairs), Russia (3,500 pairs), Sweden, Finland and Denmark (16,000 pairs), Ireland (17,000 pairs), and France (25 pairs). It is very uncertain what proportions of birds from these populations migrate through UK waters, or winter in UK waters. The biogeographic population with connectivity to UK waters comprises 1,707,000 birds (adults and immatures) with 357,000 of these from the UK population and 1,350,000 from overseas populations. Numbers estimated to be present in UK waters in the migration seasons (August to October, and January to March) are 1,197,000 birds in total, with 347,000 of these from the UK and 850,000 from overseas populations. Numbers

estimated to be present in UK waters in winter (November-December) are 559,000 birds in total, with 99,000 of these from the UK and 460,000 from overseas populations.

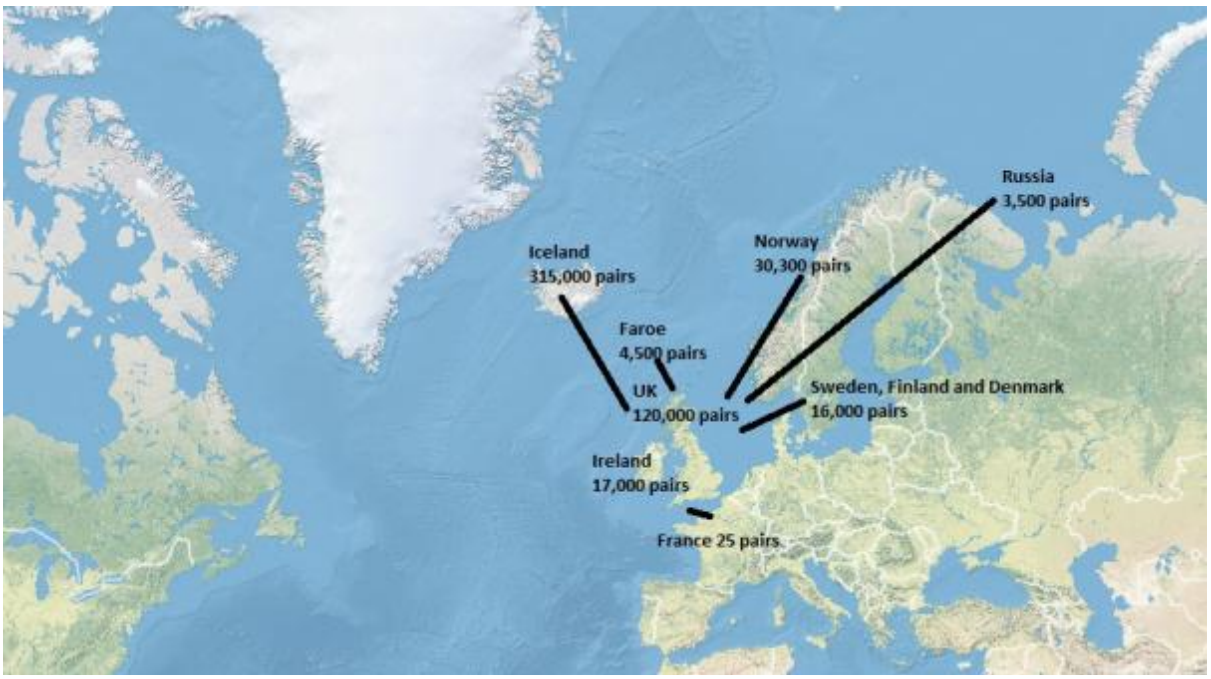


Figure 22.2. Breeding population origins of razorbills in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

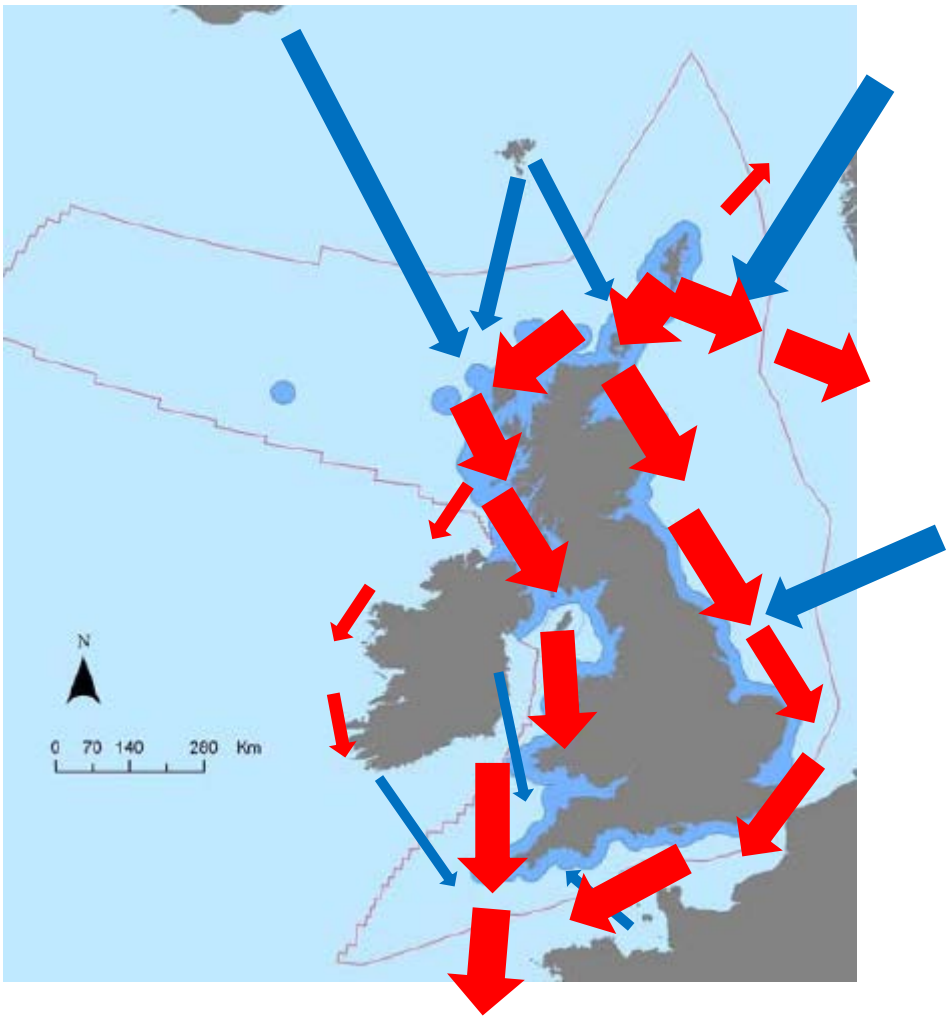


Figure 22.3. Main movements of razorbills from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

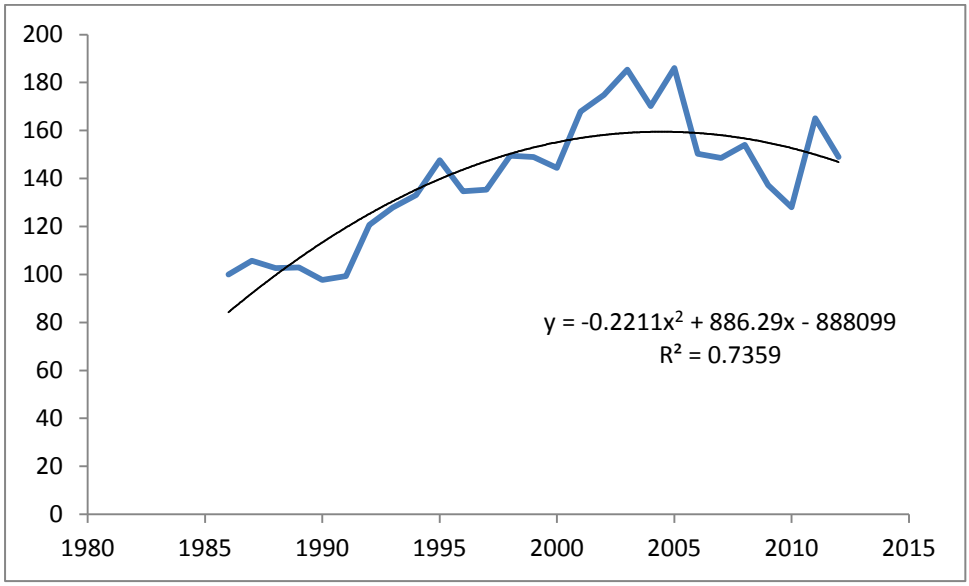


Figure 22.4. Trend in the razorbill breeding population index in UK from 1986-2012. Data from JNCC seabird population monitoring database.

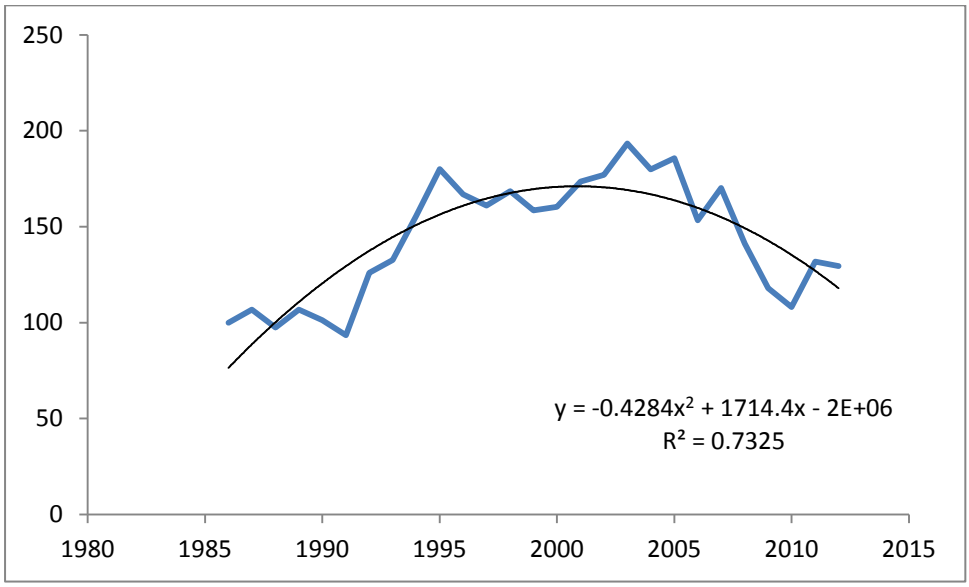


Figure 22.5. Trend in the razorbill breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

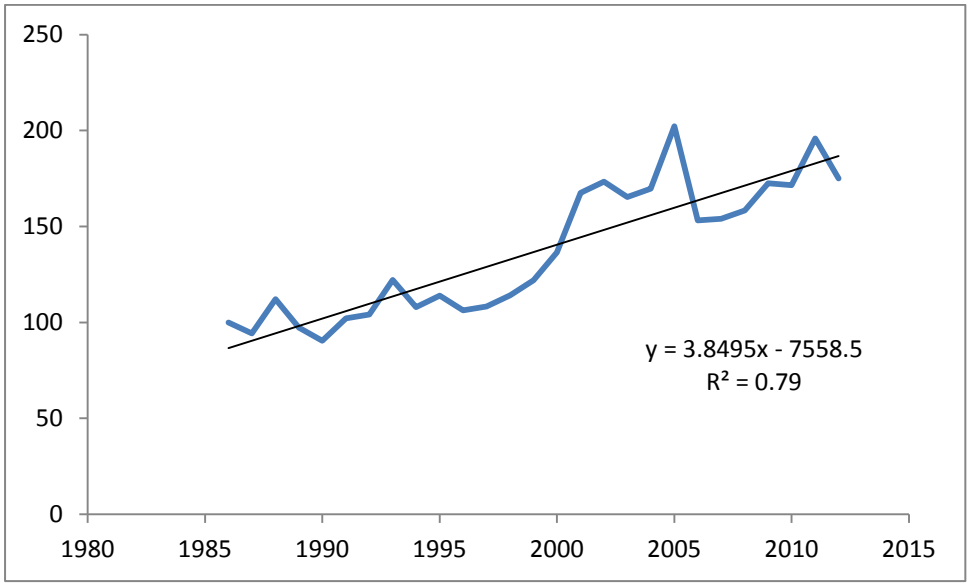


Figure 22.6. Trend in the razorbill breeding population index in Wales from 1986-2012. Data from JNCC seabird population monitoring database.

22.9 Proportion of BDMPS from UK breeding SPAs

The 19 SPAs with breeding razorbills as a feature together held 81,335 pairs at designation, estimated to represent ca. 76% of the British breeding population and ca. 26% of the all-Ireland breeding population (Stroud et al. 2001). From survey data in 1998-2011, Stroud et al. (2014) estimated that the 18 SPA populations designated in Britain held 92.9% of the British population, while the single SPA designated for razorbill in Northern Ireland held about 66% of the all-Ireland population.

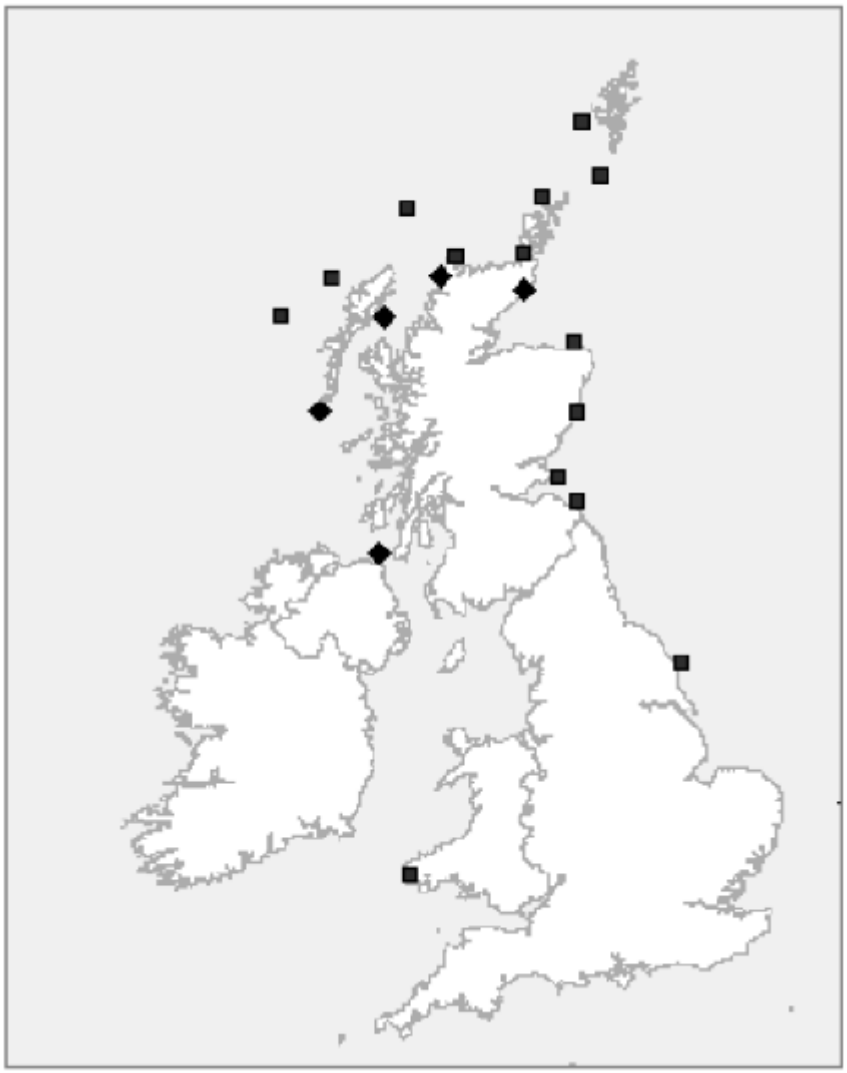


Figure 22.7. UK SPA suite for razorbill. These SPA populations are listed in Table 22.1.

Table 22.1. The UK SPA suite for breeding razorbills.

SPA	Location	Pairs	Year desig-nated	Site Condition Monitoring*	Recent count (pairs)	Year	Reference
UK North Sea & Channel							
Foula	Shetland	4,154	1995	Declined 2007	2,814 375	2000 2007	SMP database SMP database
Fair Isle	Shetland	2,044	1994	Maintained 2005	2,292 915	2005 2010	SMP database SMP database
West Westray	Orkney	1,307	1996	Maintained 2007	1,600 550	1999 2007	Seabird2000 Lewis et al. 2012
North Caithness Cliffs	N Scotland	2,212	1996	Declined 2000	1,700	2000	Seabird2000
East Caithness Cliffs	N Scotland	9,259 (1986)	1996	Maintained 1999	12,500	1999	Seabird2000
Troup, Pennan & Lion's Heads	NE Scotland	3,216 (1995)	1997	Declined 2007	3,237 1,743	2001 2007	SMP database SMP database

Fowlsheugh	NE Scotland	4,576	1992	Maintained 1999	4,263 2,868 3,103 3,524	1999 2006 2009 2012	SMP database SMP database SMP database SMP database
Forth Islands	E Scotland	1,400 (1985) Or 2,693 (Stroud et al. 2001)	1990	Maintained 2007	2,403 2,534 2,489 2,625	2009 2010 2011 2012	SMP database SMP database SMP database SMP database
St Abb's Head to Fast Castle	E Scotland	1,407	1997	Maintained 1998	1,483 1,486 1,130 1,219	1998 2003 2008 2013	SMP database SMP database SMP database SMP database
Flamborough Head & Bempton (to be subsumed into Flamborough and Filey Coast SPA)	E England	5,133 (1987)	1993		5,721 10,001	2000 2008	SMP database SMP database
Flamborough and Filey Coast	E England	10,570 (2008-2011)	Not yet				See row above
UK Western waters							
North Rona and Sula Sgeir	N Scotland	1,541 (1986)	2001	Declined 2012	1,089 North Rona only: 552 344	1998 1998 2012	SMP database SMP database SMP database
Cape Wrath	NW Scotland	1,206	1996	Maintained 2000	2,090	2000	Seabird2000
Handa	NW Scotland	10,432 (1997)	1990	Declining 2006	11,384 8,660 5,165	2001 2006 2010	SMP database SMP database SMP database
St Kilda	Western Isles	2,546	1992	Maintained 2000	1,700	1999	Seabird2000
Shiant Isles	Western Isles	7,337 (1986)	1992	Declined 2008	5,391 4,248	1999 2008	SMP database SMP database
Flannan Isles	Western Isles	2,117 (1988)	1992	Recovering 2013	1,051	1998	SMP database
Mingulay and Berneray	Western Isles	11,323 (1985)	1994	Declined 2009	15,343 22,633 10,111	1998 2003 2009	SMP database SMP database SMP database
Rathlin Island	N Ireland	5,978 (1985)	1999		13,976 7,158 15,393	1999 2007 2011	SMP database SMP database SMP database
Skomer and Skokholm	Wales	2,854 (1997)	1982		2,800 2,631 2,198 2,699 2,607 6,001	2008 2009 2010 2011 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

22.10 BDMPS

Two spatial BDMPS areas can be defined; the UK North Sea and Channel, and UK western waters. Birds from colonies in the UK North Sea tend to remain in the North Sea or to migrate south through the North Sea and Channel to reach winter quarters in southern Europe. Birds from colonies in UK western waters tend to migrate south through UK western waters, and very few from those colonies enter the North Sea. So these two BDMPS are fairly discrete populations. However, razorbills migrate further southwards than common guillemots, and relatively few razorbills from UK colonies remain in UK waters in winter, so there is a need to separate two distinct seasonal BDMPS periods; migration seasons (August-October, and January-March), and winter (November-December).

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 64 to 67.

Based on evidence reviewed in sections 22.5, 22.6 and 22.7, the UK North Sea and Channel non-breeding season BDMPS is estimated to hold 95% of adults and 90% of immatures from colonies in Shetland, Orkney and north Caithness, 100% of adults and 90% of immatures from colonies on the UK North Sea coast from Caithness to East Anglia, 2% of adults and 5% of immatures from colonies in UK western waters from NW Scotland to SW England, 5% of adults and 10% of immatures from Russia, 30% of adults and 40% of immatures from Iceland, 20% of adults and 50% of immatures from Norway, 10% of adults and 30% of immatures from Denmark, Finland and Sweden, 50% of birds from Faroe, 2% of adults and 5% of immatures from Ireland, 1% of adults and 2% of immatures from France. These proportions result in an estimated BDMPS for the migration seasons of 591,874 birds, 157,443 from UK and 434,431 from overseas populations (Appendix A Table 64).

Based on evidence reviewed in sections 22.5, 22.6 and 22.7, the UK western waters non-breeding season BDMPS is estimated to hold 5% of adults and immatures from colonies in Shetland, Orkney and north Caithness, 0% of adults and 2% of immatures from colonies on the UK North Sea coast from Caithness to East Anglia, 98% of adults and 90% of immatures from colonies in UK western waters from NW Scotland to SW England, 5% of adults and 10% of immatures from Russia, 30% of adults and 40% of immatures from Iceland, 10% of adults and 30% of immatures from Norway, 5% of adults and 10% of immatures from Denmark, Finland and Sweden, 50% of birds from Faroe, 10% of adults and immatures from Ireland, 5% of adults and immatures from France. These proportions result in an estimated BDMPS for the migration seasons of 606,914 birds, 190,035 from UK and 416,879 from overseas populations (Appendix A Table 65).

Based on evidence reviewed in sections 22.5, 22.6 and 22.7, the UK North Sea and Channel winter season BDMPS is estimated to hold 30% of adults and 10% of immatures from UK North Sea colonies, 10% of adults and 5% of immatures from colonies in UK western waters in Scotland, 5% of adults and no immatures from colonies in Northern Ireland, Wales and SW England, 1% of adults and 2% of immatures from Russia, 10% of adults and 20% of immatures from Iceland, 5% of adults and 10% of immatures from Norway, 2% of adults and 5% of immatures from Denmark, Finland and Sweden, 30% of birds from Faroe, 1% of adults and 2% of immatures from Ireland, 5% of adults and 5% of immatures from France. These proportions result in an estimated BDMPS for the winter season of 218,622 birds, 45,753 from UK and 172,869 from overseas populations (Appendix A Table 66).

Based on evidence reviewed in sections 22.5, 22.6 and 22.7, the UK western waters winter season BDMPS is estimated to hold 1% of adults and 2% of immatures from colonies in the UK North Sea coast, 40% of adults and 10% of immatures from colonies in UK western waters from NW Scotland to Northern Ireland, 30% of adults and 10% of immatures from

colonies in Wales and SW England, 1% of adults and 2% of immatures from Russia, 20% of adults and 30% of immatures from Iceland, 5% of adults and 10% of immatures from Norway, 2% of adults and 5% of immatures from Denmark, Finland and Sweden, 30% of birds from Faroe, 10% of adults and 10% of immatures from Ireland, 5% of adults and 5% of immatures from France. These proportions result in an estimated BDMPS for the winter season of 341,422 birds, 53,063 from UK and 288,359 from overseas populations (Appendix A Table 67).

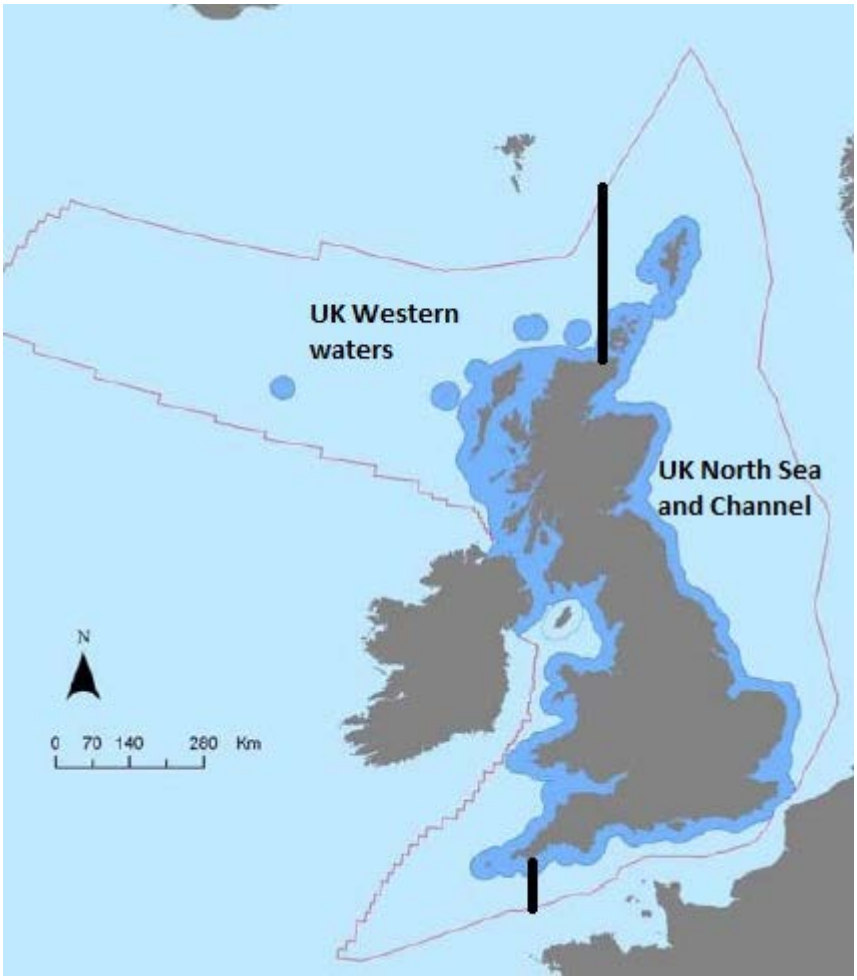


Figure 22.8. Two defined BDMPS spatial areas for razorbill: 'UK North Sea waters and Channel' and 'UK Western waters'.

22.11 Proportions of UK SPA birds in BDMPS

A very high proportion of razorbills in the UK are from UK SPA populations, estimated at around 90%. Given the large number of designated colonies and the high proportion of the total population in those sites, the proportion of the BDMPS that is from UK SPAs will mainly be determined by relative numbers of birds coming from overseas populations into these areas. Those numbers of overseas birds are very uncertain. Proportions of birds that are adults from UK SPA colonies in each BDMPS can be estimated directly from the data in Appendix A Tables 64 to 67. For example, in the UK North Sea and Channel migration seasons BDMPS (591,874 birds) there are estimated to be 71,824 adults from SPA colonies, so these represent 12% of the total birds present. In the UK western waters migration seasons BDMPS (606,914 birds) there are estimated to be 92,176 adults from SPA colonies, so these also represent 15% of the total birds present.

22.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Given the high mobility of razorbills, their relatively long distance migrations, and the large numbers of migrants from overseas passing through and wintering in UK waters, birds from UK SPA populations are likely to be very well mixed within each of the BDMPS populations in migration seasons and in winter.

23. BLACK GUILLEMOT *Cepphus grylle*

Black guillemot BDMPS is defined as the population of birds resident within a circle or buffer zone of 20 km radius around any focal site. There are 26,000 pairs in UK so 52,000 adults plus 1.32 immatures/adult. No birds from overseas populations are known to visit UK waters except as rare vagrants.

23.1 Breeding range and taxa

Black guillemot has an almost circumpolar breeding range in Arctic and sub-Arctic latitudes. There are five subspecies. *C. g. arcticus* breeds in Britain, eastern North America, southern Greenland, Denmark, SW Sweden and from Norway to the White Sea. Nominate *C. g. grylle* breeds only in the Baltic. Subspecies *faeroeensis* only in Faroe. Subspecies *islandicus* only in Iceland. Subspecies *mandtii* from northern Siberia to arctic Canada and northern Greenland. There appears to have been no assessment of whether biometrics would allow origins of individuals to be identified, but evidence indicates that hardly any birds from overseas have ever reached UK waters, so in view of the highly sedentary nature of this species within the British Isles and in nearby countries, there is unlikely to be any detectable numbers of birds from overseas reaching the UK.

23.2 Non-breeding component of the population

Black guillemots start to breed when 4 years old (BTO Birdfacts). Adult survival rate is 0.87 (BTO Birdfacts), juvenile survival unknown (BTO Birdfacts) and mean productivity is 1.295 chicks per pair (JNCC database, n=58 measurements). To obtain a stable population, survival of immatures was adjusted to 0.5 for juveniles, 0.6 for 1-year olds, 0.77 for 2-year olds, and 0.87 for 3-year olds. The model population comprised 43% adults, 28% juveniles and 29% older immatures. There are 1.32 immatures per adult.

23.3 Phenology

Breeding colonies in the UK are deserted in September, with modal departure in August (Pennington et al. 2004; Forrester et al. 2007). Black guillemots in the UK do not migrate, and rarely disperse far from their colonies. The Trektellen seawatching UK sites (predominantly in south and east England) reported only extremely low numbers of birds per hour, with no clear seasonal patterns apart from a slightly higher mean number in autumn and winter than in spring or summer, suggesting a slight post-breeding dispersal in August (Figure 23.1). However, it is noteworthy that numbers of black guillemots reported were considerably lower than even the numbers of roseate terns at these seawatching sites. Birds re-occupy colonies from late March, with modal return in early April (Pennington et al. 2004; Forrester et al. 2007).

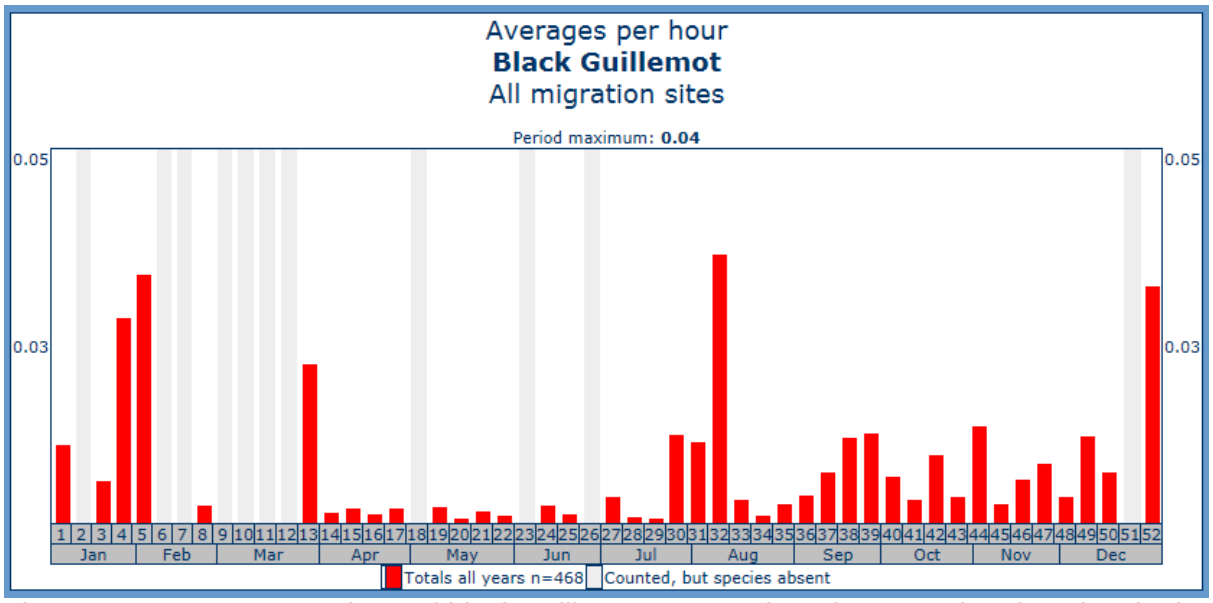


Figure 23.1. Average numbers of black guillemots counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) did not consider seasonality of black guillemot. From the data reviewed above, an appropriate definition would be breeding season April-August, non-breeding season September-March, but with negligible dispersal/migration occurring.

23.4 Defined seasons:

- UK Breeding season — April-August
- Post-breeding migration in UK waters — not evident
- **non-breeding season** — **September-March**
- Return migration through UK waters — not evident
- Migration-free breeding season — April-August
- Migration-free winter season — September-March

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for black guillemot:

Non-breeding season BDMPS (September-March).

23.5 Movements of birds from the UK population

Black guillemot populations in Britain and Ireland are considered to be sedentary, with no seasonal migration and negligible seasonal dispersal; no British ringed black guillemot has been recovered abroad (Wernham et al. 2002). However, there is evidence from counts at different times of year for birds moving away from particularly exposed coasts during winter to more sheltered coasts (e.g. from Foula and Fair Isle which are very exposed coastlines; Ewins and Kirk 1988; Pennington et al. 2004). Ring recoveries from Fair Isle include several birds that moved as far as Orkney or the north coast of Scotland and two cases where young birds were recovered in winter in east England (Wernham et al. 2002). However, Ewins and Kirk (1988) concluded that most black guillemots in Shetland never move more than 10-15 km from their natal site. Timing of such migration is difficult to assess, and probably occurs in response to severe weather so tends to occur in autumn and winter. The Trektellen seawatching data for UK, which mainly come from sites in Yorkshire, suggests that the few records of black guillemot occur mostly between late July and March, which fits in with the

idea that these birds are predominantly storm-driven juveniles seeking shelter. Shetland, Fair Isle, and Orkney Bird Reports provide very little indication of dispersal movements by black guillemots as the species is present throughout the year in those areas and there is little or no evident seasonal variation in numbers present.

23.6 Movements of birds from overseas into UK waters

Black guillemots in Faroe are also sedentary; no black guillemots ringed in Faroe have been recovered away from the archipelago (Hammer et al. 2013) and the same applies in Iceland, where only short-distance (longest documented movement 10.5 km) natal dispersal occurs (Frederiksen and Petersen 2000). The lack of movement between Iceland, Faroe and UK is also suggested by the fact that these three populations are classified into three distinct subspecies: *islandicus* in Iceland, *faeroeensis* in Faroe, and *arcticus* in UK. There are no records of *islandicus* or *faeroeensis* in Scotland (Forrester et al. 2007). Numbers of black guillemots arriving in UK waters from overseas are apparently most likely to be occasional birds from southern Norway (where there are thousands) and southern Sweden (where there are thousands) (Mitchell et al. 2004). In relation to resident populations on northern Scottish coasts (Shetland, Orkney, NE Scotland) the numbers of arrivals from Scandinavia are likely to be negligible, while along the English east coast and south-east coast of Scotland, where the species is not resident (Mitchell et al. 2004), the very small numbers that arrive there (the 2007-11 Atlas suggests some 19 records in those 5 years of survey; Balmer et al. 2013) are probably about as likely to originate from Scandinavia as from Scotland. There are two recoveries of young birds from southern Sweden recovered on the coast of east England (Wernham et al. 2002). The 2007-11 Atlas also shows 6 records in the 5 years of survey in SW England, where the species is also not resident, and those birds are likely to have originated from populations in Wales or Ireland; if exposed areas are the likely source then probably these birds moved from SW or S Ireland where there are large breeding numbers on relatively exposed coast. Apart from these very small numbers moving beyond normal breeding range, most areas hold the same population in winter as in the breeding season.

23.7 Numbers in UK waters

The black guillemot only occurs in English waters in extremely small numbers, mostly in autumn, although there are about 7 resident in Cumbria (Mitchell et al. 2014). The population in Wales is extremely small (Seabird 2000 suggested 28 resident individuals), while there are 602 at the Isle of Man (again resident so unlikely to move from there). Northern Ireland holds about 1,200 birds, Scotland about 37,000 to 38,000 birds (Mitchell et al. 2014). The Scottish population is distributed along all western and northern coasts, but is scarce in SW Scotland. In east Scotland, there are very few south of Caithness.

23.8 Biogeographic population

Stroud et al. (2001) did not define the biogeographic breeding population of this species as it is not relevant in terms of the Birds Directive because it is not migratory. However, Mitchell et al. (2004) provided an estimate of the population of the subspecies *arcticus* as 72,377-142,321 pairs. Kober et al. (2010) did not present an estimate for the biogeographic population of this species. Only UK birds and a very few from Ireland occur in UK waters, so the only populations with connectivity to UK waters are the 26,000 pairs in the UK and a very small fraction of the population in Ireland (which comprises about 2,200 pairs (Figure 23.2). No other overseas populations show significant connectivity with UK waters, although a handful of birds that reach the southern North Sea coast of England might possibly originate from Scandinavia as well as from Scotland. The biogeographic population with connectivity to UK waters can be defined as the populations of the UK and Ireland, a total of 28,200 pairs, so 56,400 adults plus 74,000 immatures. This indicates a total of 130,000 birds.



Figure 23.2. Breeding population origins of black guillemots in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

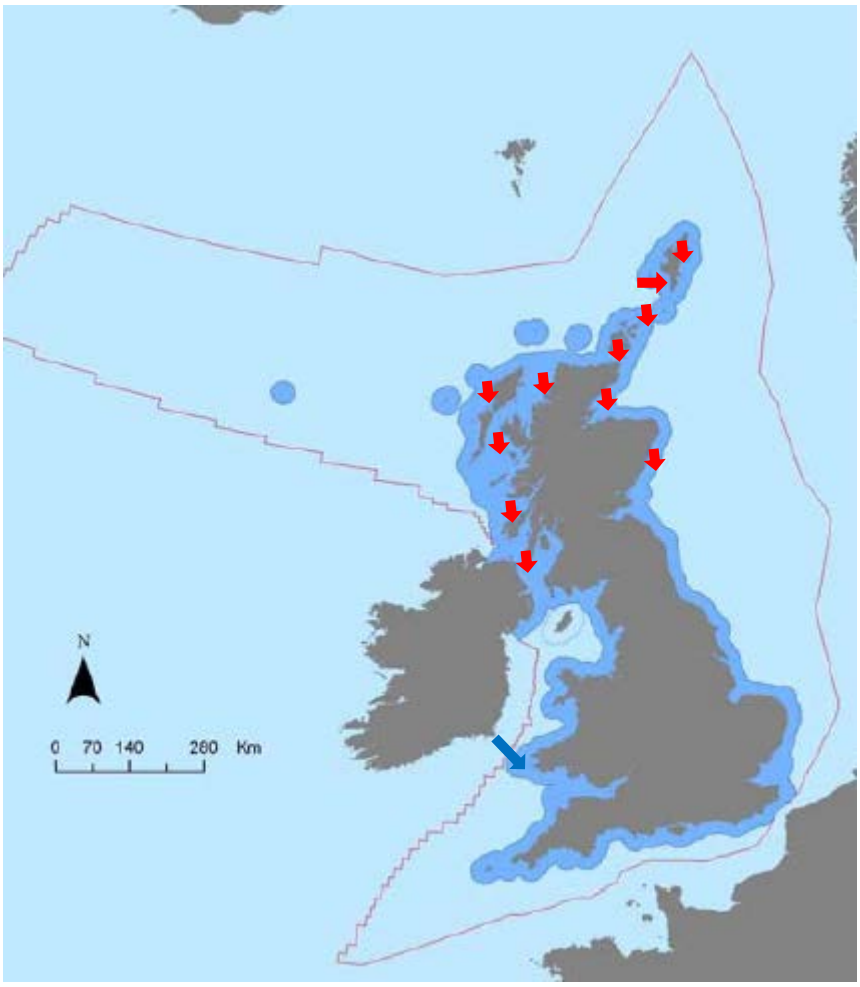


Figure 23.3. Main movements of black guillemots from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

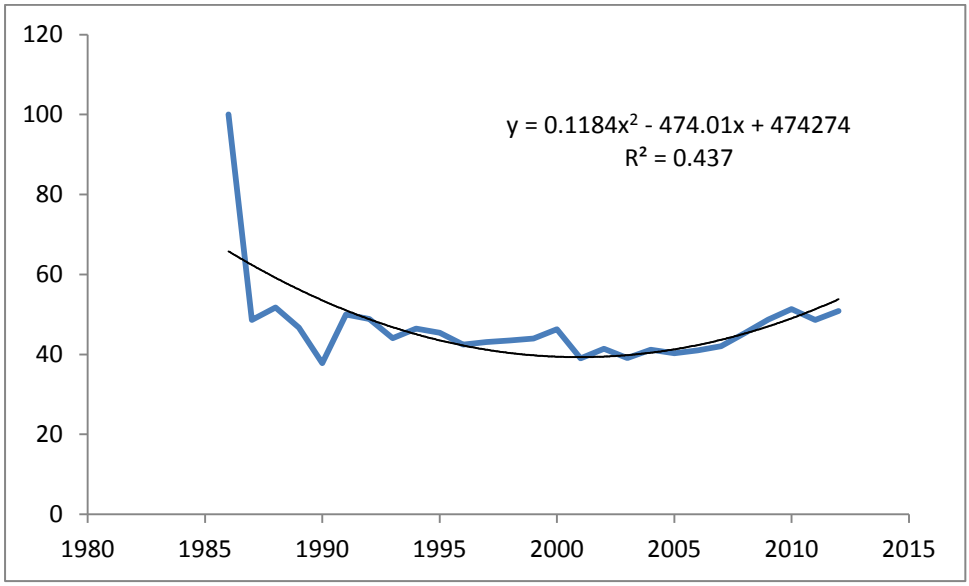


Figure 23.4. Trend in the black guillemot breeding population index in Scotland from 1986-2012. Data from JNCC seabird population monitoring database.

23.9 Proportion of UK population from UK breeding MPAs

There are no SPAs in the UK with black guillemot designated as a feature, since this species does not qualify as a migratory species. However, a number of sites are being considered for designation as Marine Protected Areas in Scottish waters with black guillemot as a designated feature. These include Clyde Sea Sill pMPA (>400 birds), East Caithness Cliffs pMPA (1,500 birds), Fetlar to Haroldswick pMPA (>2,000 birds), Monach Isles pMPA (820 birds), Papa Westray pMPA (>400 birds), and Small Isles pMPA (1,200 birds). These populations together sum to about 6,000 birds, so represent somewhere around 16% of the UK population.

23.10 BDMPS

Occasional birds that disperse exceptionally large distances (in this case exceptionally large means more than about 10-15 km) can be considered as truly exceptional. Since UK black guillemots only rarely move more than a maximum of 15 km from their natal site throughout their lifetime, almost all birds will have connectivity only with sites that are within about a 20 km radius. This allows a BDMPS to be defined as those birds found within 20 km of a specific site.

23.11 Proportions of UK MPA birds in BDMPS

This proportion will be zero for all locations except those that lie at least in part within 20 km of one of the six pMPA populations (recognising that the black guillemot feature in those pMPAs is not necessarily distributed throughout the boundary of that pMPA but may be found only in a small part of the pMPA if that is designated for multiple features rather than just for black guillemot).

23.12 Spatial distribution of UK MPA birds across the BDMPS

Within areas that overlap in their 20 km distance envelope with a pMPA black guillemot feature, the spatial distribution of MPA birds within the BDMPS is likely to be highly aggregated at the pMPA site.

24. ATLANTIC PUFFIN *Fratercula arctica*

	Biogeographic population with connectivity to UK waters (adults and immatures)	Numbers in UK waters in non-breeding season (mid-August to March) (adults and immatures)
Overseas	9,470,000	188,586
UK	2,370,000	347,928
Total	11,840,000	536,514

	Total number of birds in BDMPS (adults plus immatures)	Number from overseas populations (adults plus immatures)	Number from UK population (adults plus immatures)
Non-breeding season BDMPS (mid-August to March)			
UK North Sea and Channel	231,957	69,896	162,061
UK Western waters	304,557	118,690	185,867

Puffins are especially difficult to census because they are burrow-nesters and many of the very large colonies are partly or completely inaccessible, or in habitat where burrows cannot be identified (e.g. cliff fissures and boulder fields). Numbers of puffins are sometimes censused by counting birds on the colony surface, but such numbers fluctuate dramatically from hour to hour, day to day, and through the summer. As a result, the sizes of many puffin breeding populations are only very approximately known. This results in colour coding the estimated biogeographic population size as red. Puffins are also particularly difficult to count at sea because thyey are small, dark, spend much time underwater, and tend to dive as boats approach. So at sea surveys apparently underestimate puffin numbers. They disperse over huge areas of ocean at low densities. In addition, although large numbers have been ringed, the ring recovery rate is especially low, and probably presents a highly biased picture of where puffins die, never mind where they live during the non-breeding period. For all these reasons, the estimation of numbers of puffins in BDMPS populations is especially uncertain, so is coded red. There have been a few small projects deploying geolocators on breeding adult puffins which do provide some insights into their movements in the non-breeding season. Those studies found results that are rather divergent from the picture based on ring recovery data and at sea studies, and suggest that puffin migrations may well be changing over time in response to population density and food resources, but may also indicate large variations in behaviour between colonies, or between years. Much more deployment of geolocators, including at colonies of overseas populations would be necessary to provide higher confidence in puffin BDMPS population sizes and geographic distributions.

Calculations on which these summary tables are based, and contributions of individual SPA populations to each BDMPS, are tabulated in Appendix A Tables 68 and 69.

24.1 Breeding range and taxa

The Atlantic puffin has been split into three subspecies, nominate *arctica* in Iceland, north Norway, east Canada and most of Greenland, *naumanni* in the far north of Greenland and in Svalbard, and *grabae* in Faroe, Britain, Ireland, and southwest Norway (Wernham et al. 2002). However, the validity of these subspecies has been challenged and it is often treated as a monotypic species (e.g. Forrester et al. 2007). There is very considerable clinal variation in size, with birds from northern colonies very much larger (Barrett et al. 1985; Harris and Wanless 2011 Appendix 1). Birds from the Channel Islands have a mean wing length of 157.9 mm, while birds from Hornøya north Norway have a mean winglength of 177.6 mm and those from Spitsbergen a mean winglength over 184 mm. Such biometric variation could potentially be used to assess origins of birds sampled in winter. However, this could be complicated at a local scale where there can be significant differences in biometrics between colonies at similar latitudes. For example, puffins from St Kilda (winglength 158.2 mm) are significantly smaller than puffins from SE Scotland (winglength 161.8 mm).

24.2 Non-breeding component of the population

According to the BTO, Atlantic puffins start to breed when 5 years old (BTO Birdfacts; source of data not presented), and this value was initially used in the model, although Harris and Wanless (2011) found that the median age of first breeding on the Isle of May was at 7 years old. Adult survival rate is 0.924 (BTO Birdfacts; Harris et al. 1997), juvenile survival unknown (BTO Birdfacts) and mean productivity is 0.67 chicks per pair (JNCC database, n=94 measurements). Harris and Wanless (2011) point out that adult survival rate varied in the Isle of May population from high levels around 0.97 in the 1970s to about 0.9 in the 2000s, so adult survival is not a species-specific constant but is affected by environmental conditions. Survival rates of adults have been estimated at 0.93 in Skomer, 0.935 in Isle of May, Fair Isle, Rost and Hornoya (Harris and Wanless 2011). To obtain a stable population for a model based on the BTO data summaries, survival of adults was set at 0.924, survival of immatures was adjusted to 0.56 for juveniles, 0.66 for 1-year olds, 0.75 for 2-year olds, 0.9 for 3-year olds and 0.91 for 4-year olds. The model population comprised 55% adults, 18% juveniles and 27% older immatures. There are 0.82 immatures per adult. However, altering the age of first breeding to 7 years but retaining adult survival as 0.924 generates a model population with 1.08 immatures per adult. For the population based on Isle of May demographic data (taking average adult survival as 0.93 and age of first breeding as 7 years) there are 1.04 immatures per adult. This last scenario seems to be the most appropriate from these alternatives.

24.3 Phenology

Breeding colonies in the UK are deserted around mid-August, with modal departure in mid-July to early August (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007). Autumn dispersal/migration starts in early July (Forrester et al. 2007), late July (Pennington et al. 2004) or early August (Cramp et al. 1977-94; Wernham et al. 2002). Peak autumn migration occurs in late July (Forrester et al. 2007), early August (Pennington et al. 2004; Brown and Grice 2005), August (Wernham et al. 2002), or September-November throughout Europe (Cramp et al. 1977-94). Peak rate of change in numbers observed in autumn at Trektellen seawatching UK sites (predominantly in south and east England) occurred in July-early August, suggesting dispersal away from the coast in that period (Figure 24.1). Autumn migration is completed by August (Forrester et al. 2007), late-August (Wernham et al. 2002; Pennington et al. 2007) or December when considering the entire North Atlantic range (Cramp et al. 1977-94).

Spring migration starts in January (Cramp et al. 1977-94), February (Wernham et al. 2002; Forrester et al. 2007) or March in Shetland (Pennington et al. 2004). Peak spring migration occurs in February (Cramp et al. 1977-94), in March (Forrester et al. 2007), in March-April (Wernham et al. 2002) or in Shetland in mid-April (Pennington et al. 2004). Increase in

numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in March-April (Figure 24.1). Spring migration is completed by March (Cramp et al. 1977-94), April (Wernham et al. 2002; Forrester et al. 2007) or May in Shetland (Pennington et al. 2004).

The first spring records of Atlantic puffin in Shetland, Fair Isle, Orkney, and Argyll Bird Reports for 2007 to 2012 were from January to 24 April, but mostly in February or March, and the last records were from 23 August to 23 December, but mostly in October. Peak autumn migration was reported in July or August in most years, and peak spring migration was reported in April in most years. Birds re-occupy colonies from late February to late March, with modal return in March to mid-April (Pennington et al. 2004; Brown and Grice 2005; Forrester et al. 2007).

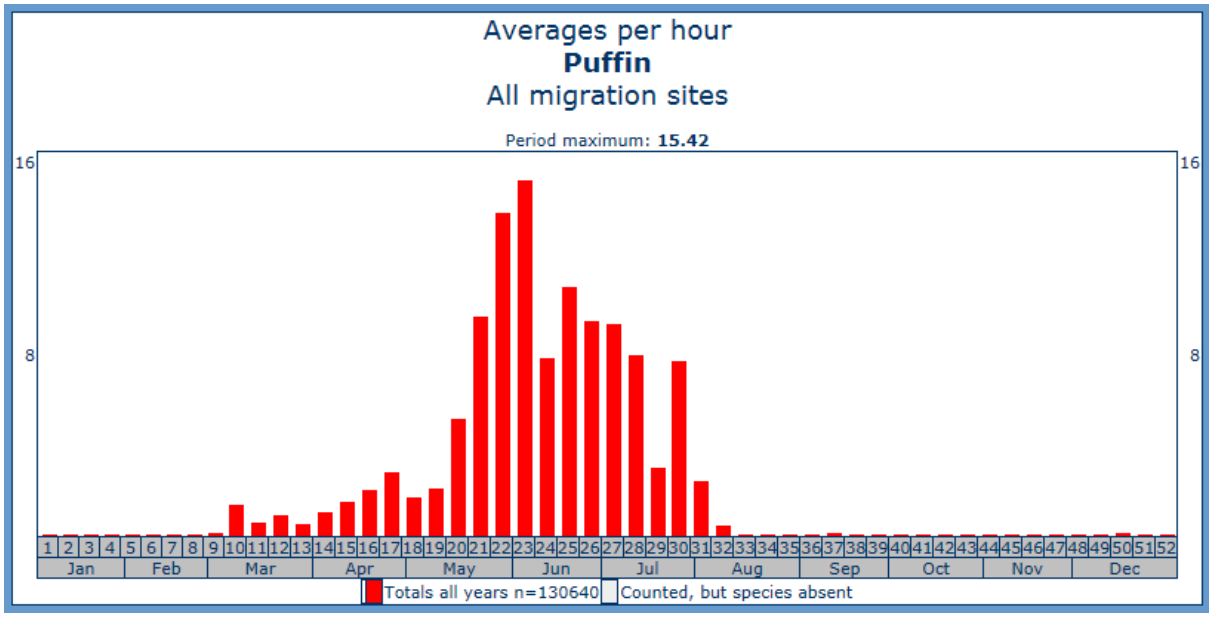


Figure 24.1. Average numbers of Atlantic puffins counted per hour at migration sites in the UK (which are mostly in south or east England). Data from Trektellen database accessed from the internet in January 2014.

Kober et al. (2010) defined breeding season as April-June, non-breeding season August-March. However, from the data reviewed above, a more appropriate definition would be breeding season April-early August, non-breeding season mid August-March.

24.4 Defined seasons:

- UK Breeding season — April-early August
- Post-breeding migration in UK waters — late July-August
- non-breeding season — mid-August-March (**non-breeding BDMPS**)
- Return migration through UK waters — March-April
- Migration-free breeding season — May-June
- Migration-free winter season — September-February

Apart from the breeding season, one seasonal BDMPS period is considered to be appropriate for Atlantic puffin:

Non-breeding season BDMPS (mid-August to March).

24.5 Movements of birds from the UK population

All puffins leave UK colonies and the immediately adjacent sea area by late August (Wernham et al. 2002; Harris and Wanless 2011). Birds apparently migrate rapidly away from breeding areas, fledglings travelling independently of adults (Harris and Wanless 2011). It used to be thought that adults undergo moult of flight feathers in mid-winter rather than immediately after chicks fledge (Harris and Yule 1977), although flightlessness in puffins due to moult has been found in all months between September and April (Harris and Wanless 2011). Recent data suggest that most adult puffins become flightless due to moult in October-November, and have generally completed renewal of primaries by December (Harris and Wanless 2011). The youngest age-classes of immature puffins apparently moult primaries in summer rather than in winter but details of how moult changes with age are rather unclear (Harris and Wanless 2011). Autumn migration takes puffins into the open sea or ocean, where they spread out thinly over huge areas. Ring recoveries come from Faroe and southern Norway to north Africa, from the western Mediterranean Sea to Newfoundland. It is thought that many puffins from UK colonies overwinter in the central North Atlantic (Wernham et al. 2002), although there is little evidence on this from ringing (Harris and Wanless 2011). All the ring recoveries of British puffins from Canadian waters were from juveniles. No adults are known from ringing to have wintered in the western North Atlantic (Wernham et al. 2002). However, geolocation data loggers deployed on breeding puffins at a colony in SW Ireland showed that most of these birds went to the Newfoundland-Labrador Shelf and remained there during August-September, moving in October back to the mid-Atlantic (Jessopp et al. 2013). This was interpreted as a strategy to exploit the abundant stock of capelin in Newfoundland waters in late summer which is seasonally concentrated in that area. That stock is the main food of puffins from local Newfoundland colonies in late summer (Hedd et al. 2010). It is possible that puffins from many colonies in the British Isles show this migration pattern, but Jessopp et al. (2013) also suggest the possibility that this might be a feature of the particular colony they studied rather than a widespread general pattern. Deployment of geolocators on breeding adult puffins at a colony in Wales (Guilford et al. 2011) also showed very rapid movement of birds westwards in August, with median positions of individuals in August from the Bay of Biscay to Newfoundland, but with most birds in an area between Newfoundland and waters south of Iceland. By October, median positions had moved to a large area between north of Iceland and west of Scotland, and by February birds were distributed widely, but much further south in an area from west of Scotland to the western Mediterranean (see figure 1 in Guilford et al. 2009). This suggests that an early migration to Newfoundland-Labrador in August may be typical for many adult puffins from British colonies, but that birds only stay in that area for a few weeks before moving eastwards, then southwards during the early winter. Guilford et al. (2009) suggest (speculatively) that this long distance but predominantly dispersive migration of puffins may be an exploratory response, rather than being based on genetic inheritance of compass instructions or cultural inheritance of traditional routes, since birds from their study colony became so widely dispersed over large areas that it is difficult to see how this would be under genetic control. Abundance of capelin in Newfoundland-Labrador waters varies enormously over the years as this is a short-lived fish which is affected by climate, and by abundance of predatory fish (especially cod) (Davoren and Montevecchi 2003; Gaston et al. 2010), so the extent to which puffins from British colonies visit Newfoundland-Labrador waters to exploit capelin in late summer may vary over years/decades as the biomass of this stock fluctuates. Ring recoveries had suggested that puffins from colonies in NE England and SE Scotland winter predominantly within the North Sea, with very few of those birds passing through the English Channel, so possibly those birds do not cross the Atlantic in the way that birds from SW Ireland and Wales have been shown to do. It had been suggested that a slight increase in numbers from North Sea colonies reaching France may reflect the increase in population size at UK North Sea colonies and so increased competition for food (Harris 1984). However, deployment of geolocators on breeding adult puffins at the Isle of May indicated that in August-December 2007 about one-third of these birds moved into the

east Atlantic, mostly off west Scotland and SW Ireland, rather than remaining in the North Sea, while most birds were distributed throughout the NW North Sea (Harris et al. 2010; 2013). This was interpreted by Harris et al. (2010) as supporting evidence from ringing that an increasing proportion of North Sea puffins were moving beyond the North Sea in response to increased population size and deteriorating conditions in the North Sea. A further deployment of geolocators on breeding adult puffins at the Isle of May in 2009 showed similar results. Interestingly, puffin survival was very poor in 2007-08 but was high in 2009-10, yet the distributions of birds overwinter in these two winters were very similar. Moving out of the North Sea into the Atlantic does not seem to correlate with over-winter survival. The geolocator data do suggest, however, that there may be substantial mixing of puffins from east and west Britain in waters west of Britain and Ireland in winter, though probably very few, if any, puffins from western colonies enter the North Sea to mix with local birds there (Harris and Wanless 2011). As with most other seabirds, ring recoveries indicate that young birds tend to travel further (south and west) from their colonies than do adults, although in the case of the puffin, the non-breeding range is not dramatically different between juveniles and adults (Wernham et al. 2002; Harris and Wanless 2011). In east Scotland, adult puffins may return to the colony in late February or March (Harris and Wanless 2011 Appendix 3), but elsewhere in the UK adults tend to return to colonies in late March or April (Wernham et al. 2002; see also Harris and Wanless 2011 Appendix 4 for Skokholm, Wales). Studies on the Isle of May indicate that about 50% of puffins reared there recruited back into that colony while 50% emigrated to breed elsewhere; birds ringed as chicks on the Isle of May have been found breeding in colonies all around the British Isles (Wernham et al. 2002). As immatures, puffins may visit several colonies before deciding where to settle to breed. These prospecting movements can take immatures to colonies hundreds of kilometres apart during the breeding season, although once a puffin has bred, which usually occurs first when 5 to 7 years old, they then remain highly faithful to their breeding site (Harris and Wanless 2011).

24.6 Movements of birds from overseas into UK waters

A total of 21 puffins ringed abroad have been recovered in the British Isles, 15 from Norway, one from Faroe, and 5 from France (Wernham et al. 2002). The one recovery of a puffin from Faroe was one of only four Faroese puffins recovered away from those islands, the others being found in France, Iceland and Greenland. So details of the migrations of Faroese puffins are unclear (Hammer et al. 2013). Although no Icelandic-ringed puffins have been recovered in the British Isles, three have been recovered in Faroe (Hammer et al. 2013) so it is reasonable to infer that some Icelandic puffins might visit UK waters during migration or winter. However, Petersen (1982, 1998) considered that SW Icelandic puffin adults most likely winter between Iceland, Greenland and Newfoundland, while those from N and E Iceland may winter from Iceland towards Norway and Faroe; there is therefore no reason to think that Icelandic puffins migrate through, or overwinter in, UK waters. According to Anker-Nilssen et al. (2000), puffins ringed in northern Norway (Barents Sea colonies) have been reported in winter from Iceland, Greenland and Newfoundland, but most recoveries have come from the southern part of the Norwegian Sea, especially around the Faroes, and in the northern part of the North Sea. However, it may be inappropriate to infer that larger numbers of recoveries in the Norwegian and North Sea imply that more puffins winter there than in the west Atlantic, since the probability of a bird being recovered may be dramatically different between these regions. Satellite tracking of five adult puffins immediately after breeding on Røst, Norway, showed all of those birds moving northwards into the Barents Sea, where densities of puffins in late summer are known to be very high (Anker-Nilssen and Aarvak 2009), suggesting that puffins from Norwegian colonies mainly disperse northwards post-breeding before moving westwards into the northern North Atlantic. Ringing data indicate that juvenile puffins from Norway are more likely to be recovered in the west Atlantic than are adults (Harris and Wanless 2011), but this may in part reflect differences in mortality risk rather than just differences in distribution between

age classes. Measurements of 98 puffins collected from beaches during a wreck in Shetland in winter 1990-91 indicated that almost all of those birds originated from colonies around the North Sea, with all age classes similarly affected; only two first-winter birds in that sample had wing lengths suggesting they came from the far north (Harris et al. 1991; Pennington et al. 2004).

24.7 Numbers in UK waters

Harris and Wanless (2011) report densities of puffins at sea in winter in the North Sea as around one bird per 20 km^2 and one bird per 5-10 km^2 in areas of the North Sea where puffins are seen regularly. European Seabirds at Sea (ESAS) data indicate a similar density in winter in waters to the west and northwest of Scotland, but somewhat lower density in waters SW of Scotland and west of Ireland, in the SE North Sea, and in the Irish and Celtic Seas (Harris and Wanless 2011). Although puffin distribution at sea in the North Sea during the breeding season reflects the distribution of colonies, birds quickly move away from colony areas in August, and form concentrations about 50 km offshore off south-east Scotland or north-east England (Harris and Wanless 2011). This concentration persists through September, but densities then decline slightly, until February-March when puffins move back to breeding sites (Harris and Wanless 2011). Fauchald and Tveraa (2009) estimated that between November and March, total numbers of puffins were 29,000 in the North Sea, 103,000 in the Norwegian Sea, and 31,000 in the Barents Sea. However, Harris and Wanless (2011) point out that if 75% of puffins from North Sea colonies are in the North Sea by January, as suggested by geolocator data from Isle of May puffins, then there should be at least 200,000 puffins in the North Sea at that time, rather than the 29,000 estimated by Fauchald and Tveraa (2009) based on the ESAS data. Harris and Wanless (2011) suggest that ESAS data may detect only about 20% or fewer of the puffins that are present, so that at-sea survey data seriously underestimate numbers of puffins dispersed over large areas of sea and ocean.

24.8 Biogeographic population

Stroud et al. (2001) defined the biogeographic breeding population as that of the subspecies *grabae* population, comprising 901,000 pairs. However, the validity of that subspecies is questionable. Mitchell et al. (2004) provided an estimate of the population of the subspecies *arcticus* (including birds of the supposed form *grabae*) as 5,500,000-6,600,000 pairs. Kober et al. (2010) presented an estimated biogeographic population of 13,500,000 individuals based on the appropriate biogeographic population being the subspecies *arcticus*. The biogeographic population with connectivity to UK waters includes populations from UK, Norway, Faroe, Ireland and France. These sum to 11,840,000 birds (adults and immatures) with 2,370,000 from UK and 9,470,000 from overseas populations. Total numbers in UK waters in the non-breeding season sum to an estimated 537,000 birds, 348,000 from the UK population and 189,000 from overseas populations, as most of the UK population moves rapidly out into the open North Atlantic across to Canada and southern Greenland rather than spending the non-breeding period in UK waters. However, we can have very little confidence in the accuracy of these estimates, and true totals may be very considerably different from these estimates. It does appear, however, that numbers present in UK waters in the non-breeding season are very small compared to the size of the biogeographic population.



Figure 24.2. Breeding population origins of puffins in UK waters during migrations and winter. Estimated numbers of breeding pairs in each population are given. Base map from OpenStreetMap www.openstreetmap.org ©OpenStreetMap contributors.

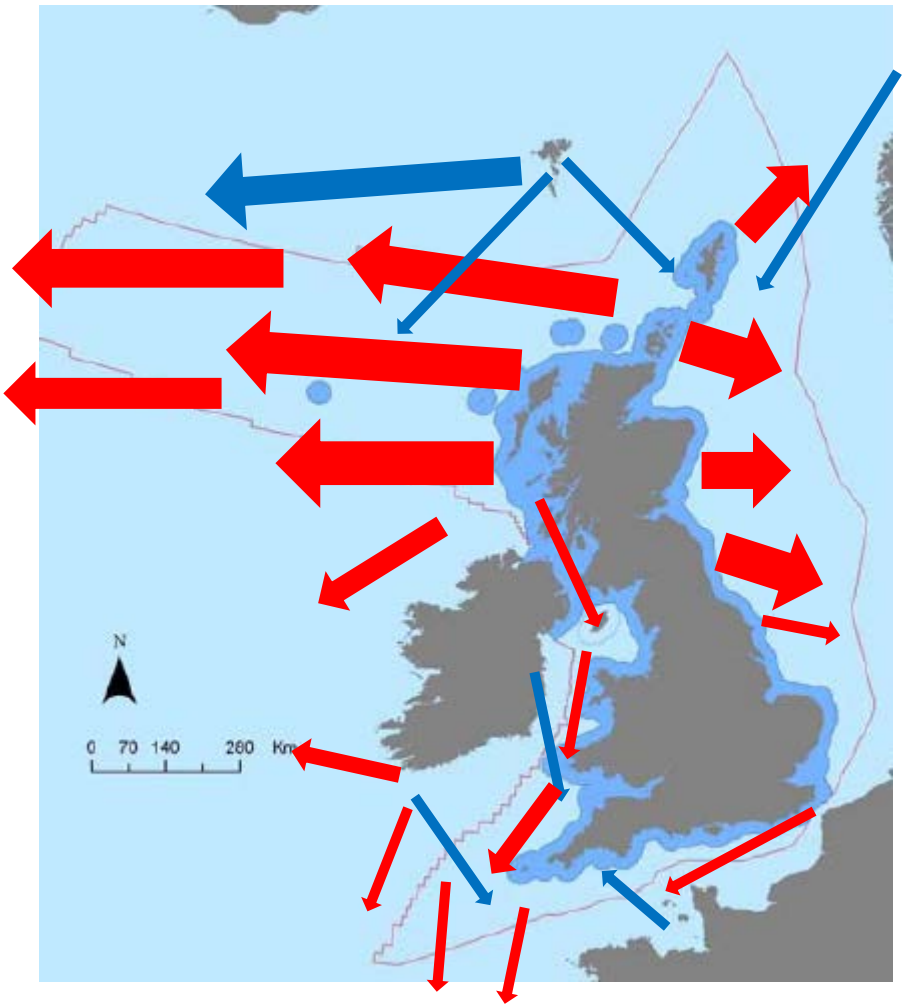


Figure 24.3. Main movements of puffins from UK breeding areas (red arrows) and from overseas populations (blue arrows) into UK waters during post-breeding dispersal/migration. Arrows imply general patterns of movement and should not be taken literally as indicating exact routes or exact starting and end points. Similarly, small numbers of birds occur in areas not marked by arrows and some birds may move in different directions from those broad patterns indicated. Movements probably tend to follow coastlines and arrows that cross land do not imply overland migration routes. As far as is known, spring return migration represents a reversal of the pattern shown in this figure.

24.9 Proportion of UK population from UK breeding SPAs

The 21 SPAs with breeding Atlantic puffins as a feature together held 470,284 pairs at designation, estimated to represent ca. 100% of the British breeding population and ca. 12% of the all-Ireland breeding population (Stroud et al. 2001). Based on survey data from 1997-2010, Stroud et al. (2014) estimated that the British SPA suite for puffin held 85.4% of the population, while the single SPA in Northern Ireland held 3.5% of the all-Ireland population.

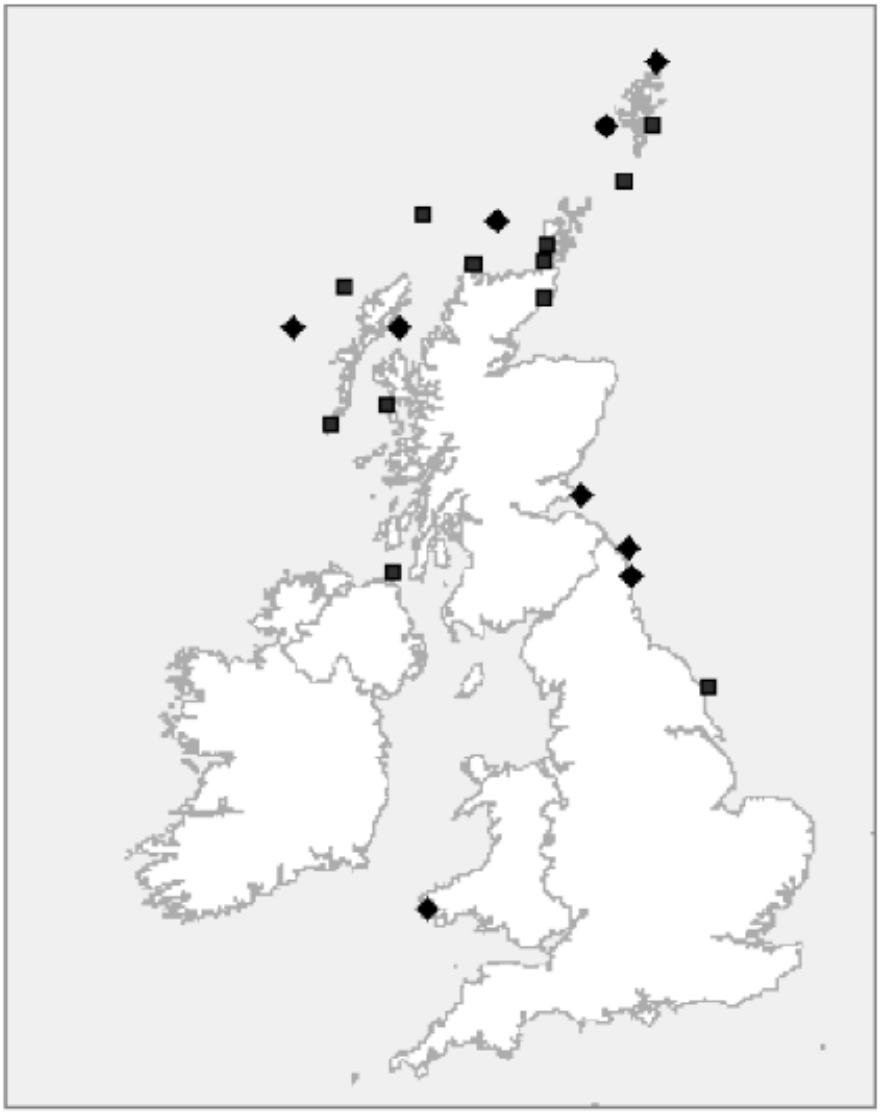


Figure 24.4. UK SPA suite for Atlantic puffin. These SPA populations are listed in Table 24.1.

Table 24.1. The UK SPA suite for breeding puffins.

SPA	Location	Pairs (or birds)	Year desig-nated	Site Condition Monitoring*	Recent count (pairs)	Year	Reference
UK North Sea & Channel							
Hermaness, Saxavord & Valla	Shetland	25,400	1994	Maintained 2002	28,300 23,661	1997 2002	Lewis et al. 2012 Lewis et al. 2012
Foula	Shetland	48,000 (1987)	1995	Declined 2007	22,500	2000	SMP database
Noss	Shetland	2,348	1996	Declined 2007	1,927 900 802	2006 2007 2007	SMP database Lewis et al. 2012 Stroud et al. 2014

Fair Isle	Shetland	8,700	1994	Declined 2009	20,244 42,500 42,000 80,000 54,000 16,700 7,278 10,706	1986 1989 1995 2000 2001 2007 2009 2012	SMP database SMP database SMP database SMP database SMP database SMP database SMP database SMP database
Hoy	Orkney	3,500	2000	Declined 2004	No recent count		No data in SMP
North Caithness Cliffs	N Scotland	(1,750) (1985-1986) in Stroud et al. 2001 but is not accurate	1996	Maintained 2000	976 7,045	2000 1999-2000	SMP database Stroud et al. 2014
East Caithness Cliffs	N Scotland	(1,750) (1985-86) in Stroud et al. 2001 but is not accurate	1996	Maintained 1999	274	1999	SMP database
Forth Islands	E Scotland	14,000 (1985) Or 21,000 (Stroud et al. 2001)	1990	Maintained 2003	21,000 62,500 83,000 50,500 62,231	1992 1998 2003 2009 2008-2010	SMP database SMP database SMP database SMP database Stroud et al. 2014
Farne Islands	NE England	34,710 (1993)	1985		55,674 36,835 39,962	2003 2008 2013	SMP database SMP database SMP database
Coquet Island	NE England	11,400 (1995)	1985		12,075 19,374 15,812 12,344	2004 2008 2009 2013	SMP database SMP database SMP database SMP database
Flamborough Head & Bempton	E England	3,473	1993		2,615 958	2000 2008	SMP database SMP database
UK Western waters							
Cape Wrath	NW Scotland	5,900	1996	Declined 2000	1,602	2000	SMP database
North Rona and Sula Sgeir	N Scotland	5,250	2001	No change 2012	5,442	2001	Mitchell et al. 2004
Sule Skerry and Sule Stack	N Scotland	43,380 (1993)	1994	Maintained 1998	59,471	1998	Seabird2000
St Kilda	Western Isles	155,000 (1989)	1992	Maintained 2000	142,264	2000	Seabird2000
Shiant Isles	Western Isles	76,100 (1970)	1992	Maintained 1999	65,170	2000	Seabird2000

Flannan Isles	Western Isles	5,500	1992	Maintained 1999	15,600	1998or 2001	SMP database
Canna and Sanday	W Scotland	1,225	1998	Maintained 1999	945	1999	SMP database
Mingulay and Berneray	Western Isles	4,000	1994	Maintained 2009	8,406 3,126	2003 2009	SMP database SMP database
Rathlin Island	N Ireland	2,398 (1985)	1999		1,579 731 695	1999 2007 2011	SMP database SMP database SMP database
Skomer and Skokholm	Wales	9,500 (mid-1980s)	1982		12,706 14,996 15,227 15,678 16,721 16,134 24,114	2003 2004 2005 2006 2007 2012 2013	SMP database SMP database SMP database SMP database SMP database SMP database SMP database

*Site Condition Monitoring data are taken from SNH Sitelink web entries for each SPA in Scotland. These data indicate the most recent formal assessment of the status of the designated feature.

24.10 BDMPS

UK birds from North Sea colonies mostly remain in winter in the North Sea, whereas western populations disperse across the North Atlantic. It is therefore appropriate to define two spatial BDMPS for puffin; UK North Sea and Channel waters, and UK western waters. Autumn dispersal is very rapid, so a single non-breeding season seems appropriate to consider, as many birds departing at the end of the breeding season spend very little time in UK waters so do not contribute to the BDMPS.

Apportioning of numbers from SPA populations, non-SPA colonies and from overseas populations is presented in Appendix A Tables 68 and 69.

Based on evidence reviewed in sections 24.5, 24.6 and 24.7, the UK North Sea and Channel non-breeding season BDMPS is estimated to hold 15% of adults and 2% of immatures from colonies in Shetland, Orkney and Caithness, 50% of adults and 2% of immatures from colonies on the east coast of the UK from Invernessshire to Humberside, 0.1% of adults and immatures from colonies in UK western waters, 0.1% of adults and 0.3% of immatures from Norway, 4% of adults and 1% of immatures from Faroe, no birds from Ireland, 5% of adults and 2% of immatures from France (Appendix A Table 68). These proportions result in an estimated non-breeding season BDMPS population of 231,957 birds, with 162,061 from the UK and 69,896 from overseas populations.

Based on evidence reviewed in sections 24.5, 24.6 and 24.7, the UK western waters non-breeding season BDMPS is estimated to hold 8% of adults and 2% of immatures from colonies in Shetland, Orkney and Caithness, 7% of adults and 2% of immatures from colonies on the east coast of the UK from Invernessshire to Humberside, 18% of adults and 2% of immatures from colonies in UK western waters, 0.2% of adults and 0.1% of immatures from Norway, 7% of adults and 2% of immatures from Faroe, 10% of birds from Ireland, 1% of birds from France (Appendix A Table 69). These proportions result in an estimated non-breeding season BDMPS population of 304,557 birds, with 185,867 from the UK and 118,690 from overseas populations.

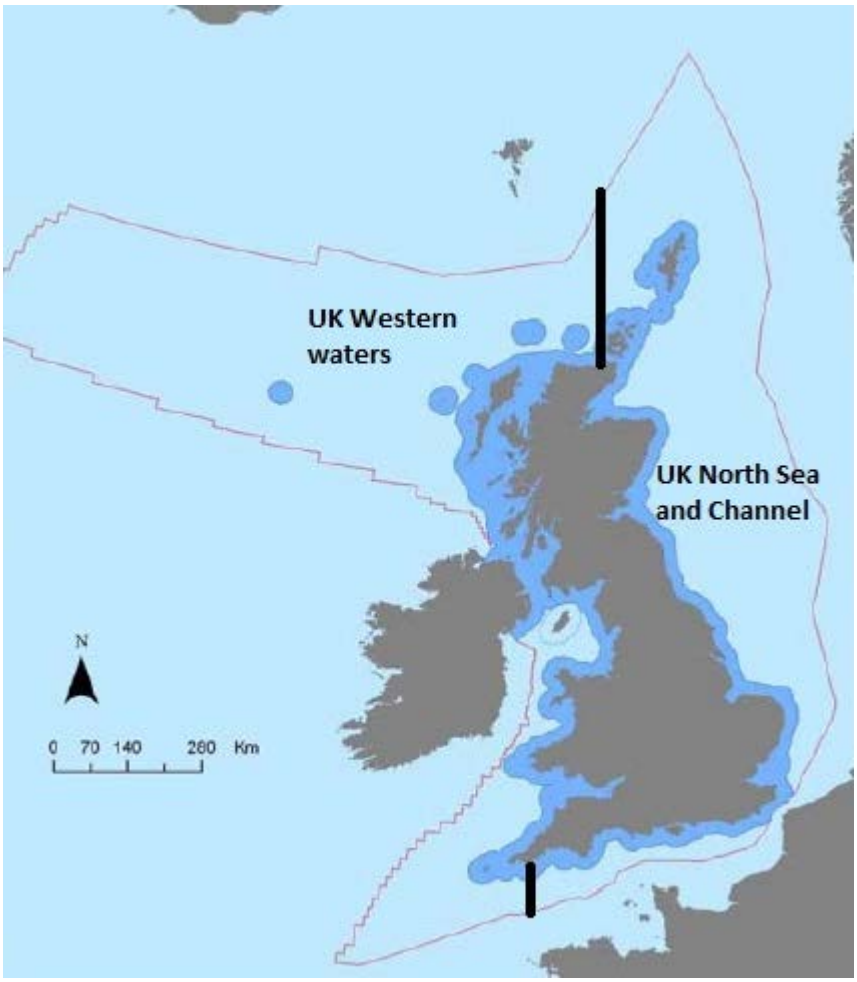


Figure 21.8. Two defined BDMPS spatial areas for Atlantic puffin: 'UK North Sea waters and Channel' and 'UK Western waters'.

24.11 Proportions of UK SPA birds in BDMPS

Proportions of birds that are adults from UK SPA colonies in each BDMPS can be estimated directly from the data in Appendix A Tables 68 and 69. For example, in the UK North Sea and Channel BDMPS (231,957 birds) there are estimated to be 134,858 adults from SPA colonies, so these represent 58% of the total birds present.

24.12 Spatial distribution of UK breeding SPA birds across the BDMPS

Given apparent high mobility of puffins, their long and rapid migrations, UK SPA birds at sea in UK waters are likely to be well mixed with birds from non-SPA colonies and from overseas.

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26. APPENDIX A.Contributions of individual SPA populations and of UK non-SPA populations and overseas populations to each BDMPS

Table 1. BDMPS for red-throated diver in winter season (December and January) in 'NW North Sea' area.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in NW North Sea waters in winter	Proportion immatures in NW North Sea waters in winter	NW N Sea Number adults	NW N Sea Number immatures	NW N Sea Total birds
Greenland	1990s	1000	2000	1480	0.05	0.05	100	74	174
Fennoscandia	1990s	5500	11000	8140	0.01	0.01	110	81	191
Hermaness, Saxavord	2013	16	32	24	0.5	0.2	16	5	21
Otterswick & Graveland	2006	25	50	37	0.5	0.2	25	7	32
Ronas Hill, North Roe	2006	50	100	74	0.5	0.2	50	15	65
Foula	2013	12	24	18	0.5	0.2	12	4	16
Orkney Mainland Moors	2007	28	56	41	0.5	0.2	28	8	36
Hoy	2007	60	120	89	0.5	0.2	60	18	78
Caithness & Sutherland	2006	46	92	68	0.5	0.2	46	14	60
Non-SPA UK North Sea	2005	600	1200	888	0.5	0.2	600	178	778
Lewis Peatlands	2006	80	160	118	0.05	0.05	8	6	14
Mointeach Scadabhaigh	2006	17	34	25	0.05	0.05	2	1	3
Rum	2013	11	22	16	0.05	0.05	1	1	2
Non-SPA UK western	2005	310	620	459	0.05	0.05	31	23	54
Overseas birds							210	155	365
UK birds							879	279	1,158
Total							1,089	434	1,523

Table 2. BDMPS for red-throated diver in winter season (December and January) in 'SW North Sea' area.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in SW North Sea waters in winter	Proportion of immatures in SW North Sea waters in winter	SW N Sea Number adults	SW N Sea Number immatures	SW N Sea Total birds
Greenland	1990s	1000	2000	1480	0.02	0.05	40	74	114
Fennoscandia	1990s	5500	11000	8140	0.4	0.6	4400	4884	9284
Hermaness, Saxavord	2013	16	32	24	0.2	0.3	6	7	14
Otterswick & Graveland	2006	25	50	37	0.2	0.3	10	11	21
Ronas Hill, North Roe	2006	50	100	74	0.2	0.3	20	22	42
Foula	2013	12	24	18	0.2	0.3	4.8	5	10
Orkney Mainland Moors	2007	28	56	41	0.2	0.3	11	12	24
Hoy	2007	60	120	89	0.2	0.3	24	27	51
Caithness & Sutherland	2006	46	92	68	0.2	0.3	18	20	39
Non-SPA UK North Sea	2005	600	1200	888	0.2	0.3	240	266	506
Lewis Peatlands	2006	80	160	118	0.05	0.05	8	6	14
Mointeach Scadabhaigh	2006	17	34	25	0.05	0.05	2	1	3
Rum	2013	11	22	16	0.05	0.05	1	1	2
Non-SPA UK western	2005	310	620	459	0.05	0.05	31	23	54
Overseas birds							4,440	4,958	9,398
UK birds							377	403	779
Total							4,817	5,361	10,177

Table 3. BDMPS for red-throated diver in winter season (December and January) in 'West of Scotland' area.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in West of Scotland waters in winter	Proportion immatures in West of Scotland waters in winter	West of Scotland Number adults	West of Scotland Number immatures	West of Scotland Total birds
Greenland	1990s	1000	2000	1480	0.02	0.05	40	74	114
Fennoscandia	1990s	5500	11000	8140	0	0.01	0	81	81
Hermaness, Saxavord	2013	16	32	24	0.05	0.1	2	2	4
Otterswick & Graveland	2006	25	50	37	0.05	0.1	2	4	6
Ronas Hill, North Roe	2006	50	100	74	0.05	0.1	5	7	12
Foula	2013	12	24	18	0.05	0.1	1	2	3
Orkney Mainland Moors	2007	28	56	41	0.05	0.1	3	4	7
Hoy	2007	60	120	89	0.05	0.1	6	9	15
Caithness & Sutherland	2006	46	92	68	0.05	0.1	5	7	11
Non-SPA UK North Sea	2005	600	1200	888	0.05	0.1	60	89	149
Lewis Peatlands	2006	80	160	118	0.4	0.2	64	24	88
Mointeach Scadabhaigh	2006	17	34	25	0.4	0.2	14	5	19
Rum	2013	11	22	16	0.4	0.2	9	3	12
Non-SPA UK western	2005	310	620	459	0.4	0.2	248	92	340
Overseas birds							40	155	195
UK birds							418	248	666
Total							458	403	861

Table 4. BDMPS for red-throated diver in winter season (December and January) in 'NW England and Wales' area.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in NW E & Wales in winter	Proportion immatures in NW E & Wales in winter	NW E & Wales Number adults	NW E & Wales Number immatures	NW E & Wales Total birds
Greenland	1990s	1000	2000	1480	0.1	0.3	200	444	644
Fennoscandia	1990s	5500	11000	8140	0.02	0.05	220	407	627
Hermaness, Saxavord	2013	16	32	24	0.02	0.05	1	1	2
Otterswick & Graveland	2006	25	50	37	0.02	0.05	1	2	3
Ronas Hill, North Roe	2006	50	100	74	0.02	0.05	2	4	6
Foula	2013	12	24	18	0.02	0.05	0	1	1
Orkney Mainland Moors	2007	28	56	41	0.02	0.05	1	2	3
Hoy	2007	60	120	89	0.02	0.05	2	4	7
Caithness & Sutherland	2006	46	92	68	0.02	0.05	2	3	5
Non-SPA UK North Sea	2005	600	1200	888	0.02	0.05	24	44	68
Lewis Peatlands	2006	80	160	118	0.2	0.2	32	24	56
Mointeach Scadabhaigh	2006	17	34	25	0.2	0.2	7	5	12
Rum	2013	11	22	16	0.2	0.2	4	3	8
Non-SPA UK western	2005	310	620	459	0.2	0.2	124	92	216
Overseas birds							420	851	1,271
UK birds							201	186	386
Total							621	1,037	1,657

Table 5. BDMPS for red-throated diver in winter season (December and January) in 'SW England and Channel' area.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in SW E & Channel in winter	Proportion immatures in SW E & Channel in winter	SW E & Channel Number adults	SW E & Channel Number immatures	SW E & Channel Total birds
Greenland	1990s	1000	2000	1480	0.1	0.2	200	296	496
Fennoscandia	1990s	5500	11000	8140	0.01	0.03	110	244	354
Hermaness, Saxavord	2013	16	32	24	0.02	0.05	1	1	2
Otterswick & Graveland	2006	25	50	37	0.02	0.05	1	2	3
Ronas Hill, North Roe	2006	50	100	74	0.02	0.05	2	4	6
Foula	2013	12	24	18	0.02	0.05	0	1	1
Orkney Mainland Moors	2007	28	56	41	0.02	0.05	1	2	3
Hoy	2007	60	120	89	0.02	0.05	2	4	7
Caithness & Sutherland	2006	46	92	68	0.02	0.05	2	3	5
Non-SPA UK North Sea	2005	600	1200	888	0.02	0.05	24	44	68
Lewis Peatlands	2006	80	160	118	0.1	0.2	16	24	40
Mointeach Scadabhaigh	2006	17	34	25	0.1	0.2	3	5	8
Rum	2013	11	22	16	0.1	0.2	2	3	5
Non-SPA UK western	2005	310	620	459	0.1	0.2	62	92	154
Overseas birds							310	540	850
UK birds							117	186	303
Total							427	726	1,153

Table 6. BDMPS for red-throated diver in migration seasons (September-November and February-April) in 'UK North Sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea waters in migration seasons	Proportion of immatures in UK North Sea waters in migration seasons	UK N Sea Number adults	UK N Sea Number immatures	UK N Sea Total birds
Greenland	1990s	1000	2000	1480	0.08	0.15	160	222	382
Fennoscandia	1990s	5500	11000	8140	0.45	0.65	4950	5291	10241
Hermaness, Saxavord	2013	16	32	24	0.95	0.8	30	19	49
Otterswick & Graveland	2006	25	50	37	0.95	0.8	48	30	77
Ronas Hill, North Roe	2006	50	100	74	0.95	0.8	95	59	154
Foula	2013	12	24	18	0.95	0.8	23	14	37
Orkney Mainland Moors	2007	28	56	41	0.95	0.8	53	33	86
Hoy	2007	60	120	89	0.95	0.8	114	71	185
Caithness & Sutherland	2006	46	92	68	0.95	0.8	87	54	142
Non-SPA UK North Sea	2005	600	1200	888	0.95	0.8	1140	710	1850
Lewis Peatlands	2006	80	160	118	0.05	0.05	8	6	14
Mointeach Scadabhaigh	2006	17	34	25	0.05	0.05	2	1	3
Rum	2013	11	22	16	0.05	0.05	1	1	2
Non-SPA UK western	2005	310	620	459	0.05	0.05	31	23	54
Overseas birds							5,110	5,513	10,623
UK birds							1,632	1,022	2,654
Total							6,742	6,535	13,277

Table 7. BDMPS for red-throated diver in migration seasons (September-November and February-April) in 'UK western waters plus Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in migration seasons	Proportion of immatures in UK western waters & Channel in migration seasons	UK western waters & Channel Number adults	UK western waters & Channel Number immatures	UK western waters & Channel Total birds
Greenland	1990s	1000	2000	1480	0.25	0.6	500	888	1388
Fennoscandia	1990s	5500	11000	8140	0.05	0.1	550	814	1364
Hermaness, Saxavord	2013	16	32	24	0.05	0.2	2	5	6
Otterswick & Graveland	2006	25	50	37	0.05	0.2	2	7	10
Ronas Hill, North Roe	2006	50	100	74	0.05	0.2	5	15	20
Foula	2013	12	24	18	0.05	0.2	1	4	5
Orkney Mainland Moors	2007	28	56	41	0.05	0.2	3	8	11
Hoy	2007	60	120	89	0.05	0.2	6	18	24
Caithness & Sutherland	2006	46	92	68	0.05	0.2	5	14	18
Non-SPA UK North Sea	2005	878	1756	1299	0.05	0.2	88	260	348
Lewis Peatlands	2006	80	160	118	0.95	0.8	152	95	247
Mointeach Scadabhaigh	2006	17	34	25	0.95	0.8	32	20	52
Rum	2013	11	22	16	0.95	0.8	21	13	34
Non-SPA UK western	2005	400	800	592	0.95	0.8	760	474	1234
Overseas birds							1,050	1,702	2,752
UK birds							878	743	1,621
Total							1,928	2,445	4,373

Table 8. BDMPS for northern fulmar in winter (November) in 'UK North Sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea waters in winter	Proportion of immatures in UK North Sea waters in winter	UK N Sea Number adults	UK N Sea Number immatures	UK N Sea Total birds
Iceland	2008	1,000,000	2000000	1240000	0.01	0.02	20000	24800	44800
Norway	1990s	386,000	772000	478640	0.01	0.02	7720	9573	17293
Faroe	1990s	600,000	1200000	744000	0.01	0.03	12000	22320	34320
Hermaness, Saxavord	2011	7,000	14000	8680	0.7	0.3	9800	2604	12404
Fetlar	2000	8,912	17824	11051	0.7	0.3	12477	3315	15792
Foula	2007	19,758	39516	24500	0.7	0.3	27661	7350	35011
Noss	2011	5,248	10496	6508	0.7	0.3	7347	1952	9299
Sumburgh Head	2009	233	466	289	0.7	0.3	326	87	413
Fair Isle	2011	29,649	59298	36765	0.7	0.3	41509	11029	52538
West Westray	2007	677	1354	839	0.7	0.3	948	252	1200
Calf of Eday	2002	1,842	3684	2284	0.7	0.3	2579	685	3264
Rousay	2009	1,030	2060	1277	0.7	0.3	1442	383	1825
Hoy	2007	19,586	39172	24287	0.7	0.3	27420	7286	34706
Copinsay	2008	1,630	3260	2021	0.7	0.3	2282	606	2888
North Caithness Cliffs	2000	14,250	28500	17670	0.7	0.3	19950	5301	25251
East Caithness Cliffs	1999	14,202	28404	17610	0.7	0.3	19883	5283	25166
Buchan Ness to Collieston	2007	1,367	2734	1695	0.7	0.3	1914	509	2422
Troup, Pennan & Lions Heads	2007	1,795	3590	2226	0.7	0.3	2513	668	3181
Fowlsheugh	2009	193	386	239	0.7	0.3	270	72	342
Forth Islands	2010	832	1664	1032	0.7	0.3	1165	310	1474
Flamborough & Filey Coast	2008	878	1756	1089	0.7	0.3	1229	327	1556
UK North Sea non-SPA	2000	129,000	258000	159960	0.7	0.3	180600	47988	228588
Cape Wrath	2000	2,115	4230	2623	0.02	0.03	85	79	163
Handa	2012	1,870	3740	2319	0.02	0.03	75	70	144
Flannan Isles	1998	7,328	14656	9087	0.02	0.03	293	273	566

North Rona & Sula Sgeir	2012	5,000	10000	6200	0.02	0.03	200	186	386
Shiant Isles	1999	4,387	8774	5440	0.02	0.03	175	163	339
St Kilda	1999	66,055	132110	81908	0.02	0.03	2642	2457	5099
Mingulay & Berneray	2009	9,046	18092	11217	0.02	0.03	362	337	698
Rathlin Island	2011	1,518	3036	1882	0.02	0.03	61	56	117
UK Western non-SPA	2000	97,000	194000	120280	0.02	0.03	3880	3608	7488
Overseas birds							39,720	56,693	96,413
UK birds							369,088	103,235	472,323
Total							408,808	159,928	568,736

Table 9. BDMPS for northern fulmar in winter (November) in 'UK western waters & Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters & Channel in winter	Proportion of immatures in UK western waters & Channel in winter	UK western waters & Channel Number adults	UK western waters & Channel Number immatures	UK western waters & Channel Total birds
Iceland	2008	1,000,000	2000000	1240000	0.01	0.02	20000	24800	44800
Norway	1990s	386,000	772000	478640	0.01	0.02	7720	9573	17293
Faroe	1990s	600,000	1200000	744000	0.01	0.03	12000	22320	34320
Hermaness, Saxavord	2011	7,000	14000	8680	0.1	0.2	1400	1736	3136
Fetlar	2000	8,912	17824	11051	0.1	0.2	1782	2210	3993
Foula	2007	19,758	39516	24500	0.1	0.2	3952	4900	8852
Noss	2011	5,248	10496	6508	0.1	0.2	1050	1302	2351
Sumburgh Head	2009	233	466	289	0.1	0.2	47	58	104
Fair Isle	2011	29,649	59298	36765	0.1	0.2	5930	7353	13283
West Westray	2007	677	1354	839	0.1	0.2	135	168	303
Calf of Eday	2002	1,842	3684	2284	0.1	0.2	368	457	825
Rousay	2009	1,030	2060	1277	0.1	0.2	206	255	461
Hoy	2007	19,586	39172	24287	0.1	0.2	3917	4857	8775
Copinsay	2008	1,630	3260	2021	0.1	0.2	326	404	730
North Caithness Cliffs	2000	14,250	28500	17670	0.1	0.2	2850	3534	6384
East Caithness Cliffs	1999	14,202	28404	17610	0.1	0.2	2840	3522	6362
Buchan Ness to Collieston	2007	1,367	2734	1695	0.1	0.2	273	339	612
Troup, Pennan & Lions Heads	2007	1,795	3590	2226	0.1	0.2	359	445	804
Fowlsheugh	2009	193	386	239	0.1	0.2	39	48	86
Forth Islands	2010	832	1664	1032	0.1	0.2	166	206	373
Flamborough & Filey Coast	2008	878	1756	1089	0.1	0.2	176	218	393
UK North Sea non-SPA	2000	129,000	258000	159960	0.1	0.2	25800	31992	57792
Cape Wrath	2000	2,115	4230	2623	0.7	0.3	2961	787	3748
Handa	2012	1,870	3740	2319	0.7	0.3	2618	696	3314
Flannan Isles	1998	7,328	14656	9087	0.7	0.3	10259	2726	12985

North Rona & Sula Sgeir	2012	5,000	10000	6200	0.7	0.3	7000	1860	8860
Shiant Isles	1999	4,387	8774	5440	0.7	0.3	6142	1632	7774
St Kilda	1999	66,055	132110	81908	0.7	0.3	92477	24572	117049
Mingulay & Berneray	2009	9,046	18092	11217	0.7	0.3	12664	3365	16030
Rathlin Island	2011	1,518	3036	1882	0.7	0.3	2125	565	2690
UK Western non-SPA	2000	97,000	194000	120280	0.7	0.3	135800	36084	171884
Overseas birds							39,720	56,693	96,413
UK birds							323,663	136,291	459,954
Total							363,383	192,984	556,367

Table 10. BDMPS for northern fulmar in migration seasons (September & October, December to March) in 'UK North sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea waters in migration	Proportion of immatures in UK North Sea waters in migration	UK N Sea Number adults	UK N Sea Number immatures	UK N Sea Total birds
Iceland	2008	1,000,000	2000000	1240000	0.02	0.04	40000	49600	89600
Norway	1990s	386,000	772000	478640	0.02	0.04	15440	19146	34586
Faroe	1990s	600,000	1200000	744000	0.02	0.06	24000	44640	68640
Hermaness, Saxavord	2011	7,000	14000	8680	0.9	0.8	12600	6944	19544
Fetlar	2000	8,912	17824	11051	0.9	0.8	16042	8841	24882
Foula	2007	19,758	39516	24500	0.9	0.8	35564	19600	55164
Noss	2011	5,248	10496	6508	0.9	0.8	9446	5206	14652
Sumburgh Head	2009	233	466	289	0.9	0.8	419	231	651
Fair Isle	2011	29,649	59298	36765	0.9	0.8	53368	29412	82780
West Westray	2007	677	1354	839	0.9	0.8	1219	672	1890
Calf of Eday	2002	1,842	3684	2284	0.9	0.8	3316	1827	5143
Rousay	2009	1,030	2060	1277	0.9	0.8	1854	1022	2876
Hoy	2007	19,586	39172	24287	0.9	0.8	35255	19429	54684
Copinsay	2008	1,630	3260	2021	0.9	0.8	2934	1617	4551
North Caithness Cliffs	2000	14,250	28500	17670	0.9	0.8	25650	14136	39786
East Caithness Cliffs	1999	14,202	28404	17610	1	0.8	28404	14088	42492
Buchan Ness to Collieston	2007	1,367	2734	1695	1	0.8	2734	1356	4090
Troup, Pennan & Lions Heads	2007	1,795	3590	2226	1	0.8	3590	1781	5371
Fowlsheugh	2009	193	386	239	1	0.8	386	191	577
Forth Islands	2010	832	1664	1032	1	0.8	1664	825	2489
Flamborough & Filey Coast	2008	878	1756	1089	1	0.8	1756	871	2627
UK North Sea non-SPA	2000	129,000	258000	159960	1	0.8	258000	127968	385968
Cape Wrath	2000	2,115	4230	2623	0	0.06	0	157	157
Handa	2012	1,870	3740	2319	0	0.06	0	139	139
Flannan Isles	1998	7,328	14656	9087	0	0.06	0	545	545

North Rona & Sula Sgeir	2012	5,000	10000	6200	0	0.06	0	372	372
Shiant Isles	1999	4,387	8774	5440	0	0.06	0	326	326
St Kilda	1999	66,055	132110	81908	0	0.06	0	4914	4914
Mingulay & Berneray	2009	9,046	18092	11217	0	0.06	0	673	673
Rathlin Island	2011	1,518	3036	1882	0	0.06	0	113	113
UK Western non-SPA	2000	97,000	194000	120280	0	0.06	0	7217	7217
Overseas birds							79,440	113,386	192,826
UK birds							494,201	270,475	764,676
Total							573,641	383,861	957,502

Table 11. BDMPS for northern fulmar in migration seasons (September & October, December to March) in 'UK western waters & Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters & Channel in migration	Proportion of immatures in UK western waters & Channel in migration	UK western waters & Channel Number adults	UK western waters & Channel Number immatures	UK western waters & Channel Total birds
Iceland	2008	1,000,000	2000000	1240000	0.02	0.04	40000	49600	89600
Norway	1990s	386,000	772000	478640	0.02	0.04	15440	19146	34586
Faroe	1990s	600,000	1200000	744000	0.02	0.06	24000	44640	68640
Hermaness, Saxavord	2011	7,000	14000	8680	0.1	0.1	1400	868	2268
Fetlar	2000	8,912	17824	11051	0.1	0.1	1782	1105	2887
Foula	2007	19,758	39516	24500	0.1	0.1	3952	2450	6402
Noss	2011	5,248	10496	6508	0.1	0.1	1050	651	1700
Sumburgh Head	2009	233	466	289	0.1	0.1	47	29	75
Fair Isle	2011	29,649	59298	36765	0.1	0.1	5930	3676	9606
West Westray	2007	677	1354	839	0.1	0.1	135	84	219
Calf of Eday	2002	1,842	3684	2284	0.1	0.1	368	228	597
Rousay	2009	1,030	2060	1277	0.1	0.1	206	128	334
Hoy	2007	19,586	39172	24287	0.1	0.1	3917	2429	6346
Copinsay	2008	1,630	3260	2021	0.1	0.1	326	202	528
North Caithness Cliffs	2000	14,250	28500	17670	0.1	0.1	2850	1767	4617
East Caithness Cliffs	1999	14,202	28404	17610	0	0.1	0	1761	1761
Buchan Ness to Collieston	2007	1,367	2734	1695	0	0.1	0	170	170
Troup, Pennan & Lions Heads	2007	1,795	3590	2226	0	0.1	0	223	223
Fowlsheugh	2009	193	386	239	0	0.1	0	24	24
Forth Islands	2010	832	1664	1032	0	0.1	0	103	103
Flamborough & Filey Coast	2008	878	1756	1089	0	0.1	0	109	109
UK North Sea non-SPA	2000	129,000	258000	159960	0	0.1	0	15996	15996
Cape Wrath	2000	2,115	4230	2623	1	0.8	4230	2098	6328
Handa	2012	1,870	3740	2319	1	0.8	3740	1855	5595

Flannan Isles	1998	7,328	14656	9087	1	0.8	14656	7269	21925
North Rona & Sula Sgeir	2012	5,000	10000	6200	1	0.8	10000	4960	14960
Shiant Isles	1999	4,387	8774	5440	1	0.8	8774	4352	13126
St Kilda	1999	66,055	132110	81908	1	0.8	132110	65527	197637
Mingulay & Berneray	2009	9,046	18092	11217	1	0.8	18092	8974	27066
Rathlin Island	2011	1,518	3036	1882	1	0.8	3036	1506	4542
UK Western non-SPA	2000	97,000	194000	120280	1	0.8	194000	96224	290224
Overseas birds							79,440	113,386	192,826
UK birds							410,601	224,767	635,368
Total							490,041	338,153	828,194

Table 12. BDMPS for Manx shearwater in migration seasons (August to early October, late March to May) in 'UK North Sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK N Sea in migration	Proportion of immatures in UK N Sea in migration	UK North Sea number of adults	UK North Sea number of immatures	UK North Sea total birds
Iceland	1990s	8500	17000	14280	0	0.001	0	14	14
Faroe	2012	25000	50000	42000	0	0.001	0	42	42
Ireland	2000	32600	65200	54768	0	0.001	0	55	55
St Kilda	1999	4802	9604	8067	0	0.01	0	81	81
Rum	2001	120000	240000	201600	0	0.01	0	2016	2016
Aberdaron Coast & Bardsey	2001	16183	32366	27187	0	0.01	0	272	272
Skomer, Skokholm & Middleh	2011	350000	700000	588000	0	0.01	0	5880	5880
UK non-SPA colonies	2000	4000	8000	6720	0.01	0.01	80	67	147
Total overseas							0	111	111
Total UK							80	8,316	8,396
Total							80	8,427	8,507

Table 13. BDMPS for Manx shearwater in migration seasons (August to early October, late March to May) in 'UK western waters & Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters & Channel in migration	Proportion of immatures in UK western waters & Channel in migration	UK western waters & Channel Number adults	UK western waters & Channel Number immatures	UK western waters & Channel Total birds
Iceland	1990s	8500	17000	14280	0.01	0.03	170	428	598
Faroe	2012	25000	50000	42000	0.01	0.03	500	1260	1760
Ireland	2000	32600	65200	54768	0.05	0.1	3260	5477	8737
St Kilda	1999	4802	9604	8067	1	0.7	9604	5647	15251
Rum	2001	120000	240000	201600	1	0.7	240000	141120	381120
Aberdaron Coast & Bardsey	2001	16183	32366	27187	1	0.7	32366	19031	51397
Skomer, Skokholm & Middleh	2011	350000	700000	588000	1	0.7	700000	411600	1111600
UK non-SPA colonies	2000	4000	8000	6720	0.8	0.6	6400	4032	10432
Total overseas							3,930	7,165	11,095
Total UK							988,370	581,430	1,569,800
Total							992,300	588,595	1,580,895

Table 14. BDMPS for northern gannet in autumn (September to November) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in autumn	Proportion immatures in UK North Sea & Channel waters in autumn	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	2010	28500	57000	46170	0.3	0.3	17100	13851	30951
Norway	2010	4500	9000	7290	0.3	0.3	2700	2187	4887
Faroe	2012	2500	5000	4050	0.3	0.3	1500	1215	2715
Hermaness, Saxavord	2008	24353	48706	39452	0.8	0.8	38965	31561	70526
Noss	2008	9767	19534	15823	0.8	0.8	15627	12658	28285
Fair Isle	2013	3924	7848	6357	0.8	0.8	6278	5086	11364
Forth Islands	2009	55482	110964	89881	1	0.9	110964	80893	191857
Flamborough & Filey	2012	11061	22122	17919	1	0.9	22122	16127	38249
UK North Sea non-SPA colonies	2004	6000	12000	9720	1	0.9	12000	8748	20748
Sule Skerry & Sule Stack	2004	4675	9350	7574	0.1	0.2	935	1515	2450
North Rona & Sula Sgeir	2004	9225	18450	14944	0.1	0.2	1845	2989	4834
St Kilda	2004	59622	119244	96588	0.1	0.2	11924	19318	31242
Ailsa Craig	2004	27130	54260	43951	0	0.1	0	4395	4395
Grassholm	2009	39292	78584	63653	0	0.1	0	6365	6365
UK western non-SPA colonies	2004	5000	10000	8100	0	0.1	0	810	810
Ireland	2004	36000	72000	58320	0	0.1	0	5832	5832
Germany	2013	632	1264	1024	0.3	0.4	379	410	789
Total overseas							21,679	23,495	45,174
Total UK							220,661	190,464	411,125
Total							242,340	213,959	456,299

Table 15. BDMPS for northern gannet in autumn (September to November) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in autumn	Proportion immatures in UK western waters in autumn	UK west Number adults	UK west Number immatures	UK west Total birds
Iceland	2010	28500	57000	46170	0.2	0.3	11400	13851	25251
Norway	2010	4500	9000	7290	0.2	0.3	1800	2187	3987
Faroe	2012	2500	5000	4050	0.2	0.3	1000	1215	2215
Hermaness, Saxavord	2008	24353	48706	39452	0.2	0.1	9741	3945	13686
Noss	2008	9767	19534	15823	0.2	0.1	3907	1582	5489
Fair Isle	2013	3924	7848	6357	0.2	0.1	1570	636	2205
Forth Islands	2009	55482	110964	89881	0	0.1	0	8988	8988
Flamborough & Filey	2012	11061	22122	17919	0	0.1	0	1792	1792
UK North Sea non-SPA cols	2004	6000	12000	9720	0	0.1	0	972	972
Sule Skerry & Sule Stack	2004	4675	9350	7574	0.9	0.7	8415	5301	13716
North Rona & Sula Sgeir	2004	9225	18450	14944	0.9	0.7	16605	10461	27066
St Kilda	2004	59622	119244	96588	0.9	0.7	107320	67611	174931
Ailsa Craig	2004	27130	54260	43951	1	0.8	54260	35160	89420
Grassholm	2009	39292	78584	63653	1	0.8	78584	50922	129506
UK western non-SPA cols	2004	4500	9000	7290	1	0.8	9000	5832	14832
Ireland	2004	36000	72000	58320	0.2	0.3	14400	17496	31896
Germany	2013	632	1264	1024	0	0	0	0	0
Total overseas							28,600	34,749	63,349
Total UK							289,401	193,204	482,605
Total							318,001	227,953	545,954

Table 16. BDMPS for northern gannet in spring (December to March) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in spring	Proportion immatures in UK North Sea & Channel waters in spring	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	2010	28500	57000	46170	0.1	0.1	5700	4617	10317
Norway	2010	4500	9000	7290	0.2	0.2	1800	1458	3258
Faroe	2012	2500	5000	4050	0.2	0.2	1000	810	1810
Hermaness, Saxavord	2008	24353	48706	39452	0.7	0.4	34094	15781	49875
Noss	2008	9767	19534	15823	0.7	0.4	13674	6329	20003
Fair Isle	2013	3924	7848	6357	0.7	0.4	5494	2543	8036
Forth Islands	2009	55482	110964	89881	0.7	0.4	77675	35952	113627
Flamborough & Filey	2012	11061	22122	17919	0.7	0.4	15485	7168	22653
UK North Sea non-SPA cols	2004	6000	12000	9720	0.7	0.4	8400	3888	12288
Sule Skerry & Sule Stack	2004	4675	9350	7574	0	0	0	0	0
North Rona & Sula Sgeir	2004	9225	18450	14944	0	0	0	0	0
St Kilda	2004	59622	119244	96588	0	0	0	0	0
Ailsa Craig	2004	27130	54260	43951	0	0	0	0	0
Grassholm	2009	39292	78584	63653	0	0	0	0	0
UK western non-SPA cols	2004	5000	10000	8100	0	0	0	0	0
Ireland	2004	36000	72000	58320	0	0.1	0	5832	5832
Germany	2013	632	1264	1024	0.3	0.3	379	307	686
Total overseas							8,879	13,024	21,903
Total UK							154,822	71,660	226,482
Total							163,701	84,684	248,385

Table 17. BDMPS for northern gannet in spring (December to March) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in spring	Proportion immatures in UK western waters in spring	UK west Number adults	UK west Number immatures	UK west Total birds
Iceland	2010	28500	57000	46170	0.2	0.2	11400	9234	20634
Norway	2010	4500	9000	7290	0.2	0.2	1800	1458	3258
Faroe	2012	2500	5000	4050	0.3	0.3	1500	1215	2715
Hermaness, Saxavord	2008	24353	48706	39452	0.3	0.3	14612	11836	26447
Noss	2008	9767	19534	15823	0.3	0.3	5860	4747	10607
Fair Isle	2013	3924	7848	6357	0.3	0.3	2354	1907	4261
Forth Islands	2009	55482	110964	89881	0.3	0.3	33289	26964	60253
Flamborough & Filey	2012	11061	22122	17919	0.3	0.3	6637	5376	12012
UK North Sea non-SPA cols	2004	6000	12000	9720	0.3	0.3	3600	2916	6516
Sule Skerry & Sule Stack	2004	4675	9350	7574	1	0.8	9350	6059	15409
North Rona & Sula Sgeir	2004	9225	18450	14944	1	0.8	18450	11956	30406
St Kilda	2004	59622	119244	96588	1	0.8	119244	77270	196514
Ailsa Craig	2004	27130	54260	43951	1	0.8	54260	35160	89420
Grassholm	2009	39292	78584	63653	1	0.8	78584	50922	129506
UK western non-SPA cols	2004	4500	9000	7290	1	0.8	9000	5832	14832
Ireland	2004	36000	72000	58320	0.3	0.3	21600	17496	39096
Germany	2013	632	1264	1024	0	0	0	0	0
Total overseas							36,300	29,403	65,703
Total UK							355,240	240,945	596,185
Total							391,540	270,348	661,888

Table 18. BDMPS for great cormorant in non-breeding season (September to March) in 'UK NW North Sea'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK NW North Sea waters in non-breeding season	Proportion of immatures in UK NW North Sea waters in non-breeding season	UK NW N Sea Number adults	UK NW N Sea Number immatures	UK NW N Sea Total birds
Denmark	1990s	40000	80000	93600	0	0.001	0	94	94
Netherlands	1990s	20000	40000	46800	0	0.0001	0	5	5
Ireland	2000	4100	8200	9594	0	0	0	0	0
France	1990s	1500	3000	3510	0	0	0	0	0
Calf of Eday	2012	181	362	424	1	1	362	424	786
East Caithness Cliffs	2013	52	104	122	1	1	104	122	226
Forth Islands	2013	80	160	187	0.6	0.5	96	94	190
UK non-SPA NW N S	2000	1200	2400	2808	0.8	0.8	1920	2246	4166
Farne Islands	2013	87	174	204	0.1	0.2	17	41	58
Abberton Reservoir	2005	216	432	505	0	0	0	0	0
UK non-SPA SW N S	2000	2200	4400	5148	0.05	0.05	220	257	477
Sheep Island NI	2013	112	224	262	0	0.001	0	0	0
UK non-SPA W Scotland	2000	2570	5140	6014	0	0.001	0	6	6
Puffin Island Wales	2013	448	896	1048	0	0.001	0	1	1
UK non-SPA Wales SW E	2000	1750	3500	4095	0	0.001	0	4	4
Overseas total							0	98	98
UK total							2,719	3,195	5,914
Total							2,719	3,293	6,012

Table 19. BDMPS for great cormorant in non-breeding season (September to March) in 'UK SW North Sea & Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK SW North Sea & Channel waters in non-breeding season	Proportion of immatures in UK SW North Sea & Channel waters in non-breeding season	UK SW N Sea & Channel Number adults	UK SW N Sea & Channel Number immatures	UK SW N Sea & Channel Total birds
Denmark	1990s	40000	80000	93600	0	0.005	0	468	468
Netherlands	1990s	20000	40000	46800	0.001	0.01	40	468	508
Ireland	2000	4100	8200	9594	0	0.01	0	96	96
France	1990s	1500	3000	3510	0	0.01	0	35	35
Calf of Eday	2012	181	362	424	0	0	0	0	0
East Caithness Cliffs	2013	52	104	122	0	0	0	0	0
Forth Islands	2013	80	160	187	0.4	0.5	64	94	158
UK non-SPA NW N S	2000	1200	2400	2808	0.2	0.2	480	562	102
Farne Islands	2013	87	174	204	0.9	0.8	157	163	319
Abberton Reservoir	2005	216	432	505	0.8	0.7	346	354	699
UK non-SPA SW N S	2000	2200	4400	5148	0.8	0.7	3520	3604	7124
Sheep Island NI	2013	112	224	262	0	0.001	0	0	0
UK non-SPA W Scotland	2000	2570	5140	6014	0	0.001	0	6	6
Puffin Island Wales	2013	448	896	1048	0	0.001	0	1	1
UK non-SPA Wales SW E	2000	1750	3500	4095	0	0.001	0	4	4
Overseas total							40	1,067	1,107
UK total							4,566	4,787	9,353
Total							4,606	5,854	10,460

Table 20. BDMPS for great cormorant in non-breeding season (September to March) in 'UK West of Scotland waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK West of Scotland waters in non-breeding season	Proportion of immatures in UK west of Scotland waters in non-breeding season	UK West of Scotland Number adults	UK west of Scotland Number immatures	UK west of Scotland Total birds
Denmark	1990s	40000	80000	93600	0	0.0005	0	47	47
Netherlands	1990s	20000	40000	46800	0	0	0	0	0
Ireland	2000	4100	8200	9594	0	0.001	0	10	10
France	1990s	1500	3000	3510	0	0	0	0	0
Calf of Eday	2012	181	362	424	0	0	0	0	0
East Caithness Cliffs	2013	52	104	122	0	0	0	0	0
Forth Islands	2013	80	160	187	0	0	0	0	0
UK non-SPA NW N S	2000	1200	2400	2808	0	0	0	0	0
Farne Islands	2013	87	174	204	0	0	0	0	0
Abberton Reservoir	2005	216	432	505	0	0	0	0	0
UK non-SPA SW N S	2000	2200	4400	5148	0	0	0	0	0
Sheep Island NI	2013	112	224	262	0.8	0.6	179	157	336
UK non-SPA W Scotland	2000	2570	5140	6014	0.7	0.5	3598	3007	6605
Puffin Island Wales	2013	448	896	1048	0	0.01	0	10	10
UK non-SPA Wales SW E	2000	1750	3500	4095	0	0.01	0	41	41
Overseas total							0	56	56
UK total							3,777	3,216	6,993
Total							3,777	3,272	7,049

Table 21. BDMPS for great cormorant in non-breeding season (September to March) in 'UK Wales & SW England waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in Wales & SW E waters in non-breeding season	Proportion immatures in Wales & SW E waters in non-breeding season	Wales & SW E Number adults	Wales & SW E Number immatures	Wales & SW E Total birds
Denmark	1990s	40000	80000	93600	0	0.0001	0	9	9
Netherlands	1990s	20000	40000	46800	0	0.0001	0	5	5
Ireland	2000	4100	8200	9594	0	0.02	0	192	192
France	1990s	1500	3000	3510	0	0.001	0	4	4
Calf of Eday	2012	181	362	424	0	0	0	0	0
East Caithness Cliffs	2013	52	104	122	0	0	0	0	0
Forth Islands	2013	80	160	187	0	0	0	0	0
UK non-SPA NW N S	2000	1200	2400	2808	0	0	0	0	0
Farne Islands	2013	87	174	204	0	0	0	0	0
Abberton Reservoir	2005	216	432	505	0	0	0	0	0
UK non-SPA SW N S	2000	2200	4400	5148	0	0	0	0	0
Sheep Island NI	2013	112	224	262	0.2	0.4	45	105	150
UK non-SPA W Scotland	2000	2570	5140	6014	0.3	0.5	1542	3007	4549
Puffin Island Wales	2013	448	896	1048	0.6	0.4	538	419	957
UK non-SPA Wales SW E	2000	1750	3500	4095	0.6	0.4	2100	1638	3738
Overseas total							0	209	209
UK total							4,224	5,169	9,393
Total							4,224	5,378	9,602

Table 22. BDMPS for European shag in non-breeding season (September to January) in 'UK NW North Sea'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK NW North Sea in non-breeding season	Proportion immatures in UK NW North Sea in non-breeding season	UK NW N Sea Number adults	UK NW N Sea Number immatures	UK NW N Sea Total birds
Hermaness, Saxavord	2002	41	82	107	1	1	82	107	189
Foula	2013	200	400	524	1	1	400	524	924
Fair Isle	2013	204	408	534	1	1	408	534	942
East Caithness Cliffs	1999	1056	2112	2767	1	1	2112	2767	4879
Buchan Ness to Collieston	2007	331	662	867	1	1	662	867	1529
Forth Islands	2013	850	1700	2227	1	0.9	1700	2004	3704
St Abbs Head to Fast Castle	2011	160	320	419	1	0.8	320	335	655
UK NW N Sea non-SPA	2000	6000	12000	15720	1	1	12000	15720	27720
Farne Islands	2013	582	1164	1525	0.3	0.4	349	610	959
UK SW N Sea non-SPA	2000	500	1000	1310	0	0	0	0	0
Sule Skerry & Sule Stack	2011	200	400	524	0	0	0	0	0
Shiant Islands	1999	506	1012	1326	0	0	0	0	0
Canna & Sanday	2013	255	510	668	0	0	0	0	0
Mingulay & Berneray	2009	115	230	301	0	0	0	0	0
UK West of Scotld non-SPA	2000	7000	14000	18340	0	0	0	0	0
Isles of Scilly	2006	1296	2592	3396	0	0	0	0	0
UK Wales & SW E non-SPA	2000	1500	3000	3930	0	0	0	0	0
Ireland	2000	2000	4000	5240	0	0	0	0	0
Overseas total							0	0	0
UK total							18,033	23,469	41,503
Total							18,033	23,469	41,503

Table 23. BDMPS for European shag in non-breeding season (September to January) in 'UK SW North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK SW North Sea & Channel waters in non-breeding season	Proportion of immatures in UK SW North Sea & Channel waters in non-breeding season	UK SW N Sea & Channel Number adults	UK SW N Sea & Channel Number immatures	UK SW N Sea & Channel Total birds
Hermaness, Saxavord	2002	41	82	107	0	0	0	0	0
Foula	2013	200	400	524	0	0	0	0	0
Fair Isle	2013	204	408	534	0	0	0	0	0
East Caithness Cliffs	1999	1056	2112	2767	0	0	0	0	0
Buchan Ness to Collieston	2007	331	662	867	0	0	0	0	0
Forth Islands	2013	850	1700	2227	0	0.1	0	223	223
St Abbs Head to Fast Castle	2011	160	320	419	0	0.2	0	84	84
UK NW N Sea non-SPA	2000	6000	12000	15720	0	0	0	0	0
Farne Islands	2013	582	1164	1525	0.7	0.6	815	915	1730
UK SW N Sea non-SPA	2000	500	1000	1310	1	1	1000	1310	2310
Sule Skerry & Sule Stack	2011	200	400	524	0	0	0	0	0
Shiant Islands	1999	506	1012	1326	0	0	0	0	0
Canna & Sanday	2013	255	510	668	0	0	0	0	0
Mingulay & Berneray	2009	115	230	301	0	0	0	0	0
UK West of Scotland non-SPA	2000	7000	14000	18340	0	0	0	0	0
Isles of Scilly	2006	1296	2592	3396	0	0	0	0	0
UK Wales & SW E non-SPA	2000	1500	3000	3930	0	0	0	0	0
Ireland	2000	2000	4000	5240	0	0	0	0	0
Overseas total							0	0	0
UK total							1,815	2,531	4,346
Total							1,815	2,531	4,346

Table 24. BDMPS for European shag in non-breeding season (September to January) in 'UK West of Scotland waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK west of Scotland waters in non-breeding season	Proportion of immatures in UK west of Scotland waters in non-breeding season	UK west of Scotland Number adults	UK west of Scotland Number immatures	UK west of Scotland Total birds
Hermaness, Saxavord	2002	41	82	107	0	0	0	0	0
Foula	2013	200	400	524	0	0	0	0	0
Fair Isle	2013	204	408	534	0	0	0	0	0
East Caithness Cliffs	1999	1056	2112	2767	0	0	0	0	0
Buchan Ness to Collieston	2007	331	662	867	0	0	0	0	0
Forth Islands	2013	850	1700	2227	0	0	0	0	0
St Abbs Head to Fast Castle	2011	160	320	419	0	0	0	0	0
UK NW N Sea non-SPA	2000	6000	12000	15720	0	0	0	0	0
Farne Islands	2013	582	1164	1525	0	0	0	0	0
UK SW N Sea non-SPA	2000	500	1000	1310	0	0	0	0	0
Sule Skerry & Sule Stack	2011	200	400	524	1	1	400	524	924
Shiant Islands	1999	506	1012	1326	1	1	1012	1326	2338
Canna & Sanday	2013	255	510	668	1	1	510	668	1178
Mingulay & Berneray	2009	115	230	301	1	1	230	301	531
UK West of Scotland non-SPA	2000	7000	14000	18340	1	1	14000	18340	32340
Isles of Scilly	2006	1296	2592	3396	0	0	0	0	0
UK Wales & SW E non-SPA	2000	1500	3000	3930	0	0	0	0	0
Ireland	2000	2000	4000	5240	0	0.01	0	52	52
Overseas total							0	52	52
UK total							16,152	21,159	37,311
Total							16,152	21,211	37,363

Table 25. BDMPS for European shag in non-breeding season (September to January) in 'UK Wales & SW England waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in Wales & SW E waters in non-breeding season	Proportion immatures in Wales & SW E waters in non-breeding season	UK Wales & SW E Number adults	UK Wales & SW E Number immatures	UK Wales & SW E Total birds
Hermaness, Saxavord	2002	41	82	107	0	0	0	0	0
Foula	2013	200	400	524	0	0	0	0	0
Fair Isle	2013	204	408	534	0	0	0	0	0
East Caithness Cliffs	1999	1056	2112	2767	0	0	0	0	0
Buchan Ness to Collieston	2007	331	662	867	0	0	0	0	0
Forth Islands	2013	850	1700	2227	0	0	0	0	0
St Abbs Head to Fast Castle	2011	160	320	419	0	0	0	0	0
UK NW N Sea non-SPA	2000	6000	12000	15720	0	0	0	0	0
Farne Islands	2013	582	1164	1525	0	0	0	0	0
UK SW N Sea non-SPA	2000	500	1000	1310	0	0	0	0	0
Sule Skerry & Sule Stack	2011	200	400	524	0	0	0	0	0
Shiant Islands	1999	506	1012	1326	0	0	0	0	0
Canna & Sanday	2013	255	510	668	0	0	0	0	0
Mingulay & Berneray	2009	115	230	301	0	0	0	0	0
UK West of Scotland non-SPA	2000	7000	14000	18340	0	0	0	0	0
Isles of Scilly	2006	1296	2592	3396	1	1	2592	3396	5988
UK Wales & SW E non-SPA	2000	1500	3000	3930	1	1	3000	3930	6930
Ireland	2000	2000	4000	5240	0	0.03	0	157	157
Overseas total							0	157	157
UK total							5,592	7,326	12,918
Total							5,592	7,483	13,075

Table 26. BDMPS for Arctic skua in autumn migration season (August to October) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in autumn	Proportion of immatures in UK North Sea & Channel waters in autumn	UK N Sea & Channel waters Number adults	UK N Sea & Channel waters Number immatures	UK N Sea & Channel waters Total birds
High Arctic tundra	1990s	50,000	100000	71000	0.01	0.01	1000	710	1710
Iceland	1990s	7500	15000	10650	0.02	0.02	300	213	513
Fennoscandia	1990s	8000	16000	11360	0.1	0.1	1600	1136	2736
Faroe	2012	750	1500	1065	0.1	0.1	150	106	256
Fetlar	2002	83	166	118	0.6	0.4	100	47	147
Foula	2013	35	70	50	0.6	0.4	42	20	62
Fair Isle	2013	19	38	27	0.6	0.4	23	11	34
West Westray	2010	27	54	38	0.6	0.4	32	15	48
Papa Westray	2012	22	44	31	0.6	0.4	26	12	39
Hoy	2010	12	24	17	0.6	0.4	14	7	21
Rousay	2010	37	74	53	0.6	0.4	44	21	65
UK non-SPA North Sea colonies	2000*	450	900	639	0.6	0.4	540	256	796
UK non-SPA western colonies	2000*	200	400	284	0	0	0	0	0
Total overseas							3,050	2,166	5,216
Total UK							822	389	1,211
Total							3,872	2,555	6,427

*updated to 2012 using trend reported in Foster and Marrs (2012)

Table 27. BDMPS for Arctic skua in autumn migration season (August to October) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in autumn	Proportion of immatures in UK western waters in autumn	UK western Number adults	UK western Number immatures	UK western Total birds
High Arctic tundra	1990s	50,000	100000	71000	0.01	0.01	1000	710	1710
Iceland	1990s	7500	15000	10650	0.02	0.02	300	213	513
Fennoscandia	1990s	8000	16000	11360	0.05	0.05	800	568	1368
Faroe	2012	750	1500	1065	0.1	0.1	150	106	256
Fetlar	2002	83	166	118	0.4	0.3	66	35	102
Foula	2013	35	70	50	0.4	0.3	28	15	43
Fair Isle	2013	19	38	27	0.4	0.3	15	8	23
West Westray	2010	27	54	38	0.4	0.3	22	12	33
Papa Westray	2012	22	44	31	0.4	0.3	18	9	27
Hoy	2010	12	24	17	0.4	0.3	10	5	15
Rousay	2010	37	74	53	0.4	0.3	30	16	45
UK non-SPA North Sea colonies	2000*	450	900	639	0.4	0.3	360	192	552
UK non-SPA western colonies	2000*	200	400	284	1	0.7	400	199	599
Total overseas							2,250	1,598	3,848
Total UK							948	491	1,439
Total							3,198	2,089	5,287

*updated to 2012 using trend reported in Foster and Marrs (2012)

Table 28. BDMPS for Arctic skua in spring migration season (April-May) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in spring	Proportion of immatures in UK North Sea & Channel waters in spring	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
High Arctic tundra	1990s	50,000	100000	71000	0.002	0.001	200	71	271
Iceland	1990s	7500	15000	10650	0.005	0.001	75	11	86
Fennoscandia	1990s	8000	16000	11360	0.01	0.005	160	57	217
Faroe	2012	750	1500	1065	0.005	0.001	8	1	9
Fetlar	2002	83	166	118	0.4	0.1	66	12	78
Foula	2013	35	70	50	0.4	0.1	28	5	33
Fair Isle	2013	19	38	27	0.4	0.1	15	3	18
West Westray	2010	27	54	38	0.4	0.1	22	4	25
Papa Westray	2012	22	44	31	0.4	0.1	18	3	21
Hoy	2010	12	24	17	0.4	0.1	10	2	11
Rousay	2010	37	74	53	0.4	0.1	30	5	35
UK non-SPA North Sea colonies	2000*	450	900	639	0.4	0.1	360	64	424
UK non-SPA western colonies	2000*	200	400	284	0	0	0	0	0
Total overseas							442	140	582
Total UK							548	97	645
Total							990	237	1,227

*updated to 2012 using trend reported in Foster and Marrs (2012)

Table 29. BDMPS for Arctic skua in spring migration season (April-May) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in spring	Proportion of immatures in UK western waters in spring	UK western Number adults	UK western Number immatures	UK western Total birds
High Arctic tundra	1990s	50,000	100000	71000	0.01	0.01	1000	710	1710
Iceland	1990s	7500	15000	10650	0.01	0.01	150	106	256
Fennoscandia	1990s	8000	16000	11360	0.05	0.03	800	341	1141
Faroe	2012	750	1500	1065	0.05	0.02	75	21	96
Fetlar	2002	83	166	118	0.6	0.5	100	59	159
Foula	2013	35	70	50	0.6	0.5	42	25	67
Fair Isle	2013	19	38	27	0.6	0.5	23	13	36
West Westray	2010	27	54	38	0.6	0.5	32	19	52
Papa Westray	2012	22	44	31	0.6	0.5	26	16	42
Hoy	2010	12	24	17	0.6	0.5	14	9	23
Rousay	2010	37	74	53	0.6	0.5	44	26	71
UK non-SPA North Sea colonies	2000*	450	900	639	0.6	0.5	540	320	860
UK non-SPA western colonies	2000*	200	400	284	1	0.7	400	199	599
Total overseas							2,025	1,179	3,204
Total UK							1,222	685	1,907
Total							3,247	1,864	5,111

*updated to 2012 using trend reported in Foster and Marrs (2012)

Table 30. BDMPS for great skua in autumn migration season (August to October) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in autumn	Proportion of immatures in UK North Sea & Channel waters in autumn	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	1980s	5400	10800	15336	0.1	0.05	1080	767	1847
Norway	2010	360	720	1022	0.1	0.05	72	51	123
Faroe	2012	500	1000	1420	0.1	0.05	100	71	171
Hermaness, Saxavord	2013	979	1958	2780	0.6	0.3	1175	834	2009
Fetlar	2002	585	1170	1661	0.6	0.3	702	498	1200
Ronas Hill, North Roe	2002	189	378	537	0.6	0.3	227	161	388
Foula	2007	1657	3314	4706	0.6	0.3	1988	1412	3400
Noss	2013	465	930	1321	0.6	0.3	558	396	954
Fair Isle	2013	266	532	755	0.6	0.3	319	227	546
Hoy	2010	1346	2692	3823	0.6	0.3	1615	1147	2762
UK Non-SPA North Sea colonies	2000	3000	6000	8520	0.6	0.3	3600	2556	6156
Handa	2013	135	270	383	0	0	0	0	0
St Kilda	2012	181	362	514	0	0	0	0	0
UK Non-SPA western colonies	2000	100	200	284	0	0	0	0	0
Total overseas							1,252	889	2,141
Total UK							10,184	7,231	17,415
Total							11,436	8,120	19,556

Table 31. BDMPS for great skua in autumn migration season (August to October) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in autumn	Proportion of immatures in UK western waters in autumn	UK western Number adults	UK western Number immatures	UK western Total birds
Iceland	1980s	5400	10800	15336	0.2	0.05	2160	767	2927
Norway	2010	360	720	1022	0.1	0.05	72	51	123
Faroe	2012	500	1000	1420	0.3	0.05	300	71	371
Hermaness, Saxavord	2013	979	1958	2780	0.4	0.2	783	556	1339
Fetlar	2002	585	1170	1661	0.4	0.2	468	332	800
Ronas Hill, North Roe	2002	189	378	537	0.4	0.2	151	107	259
Foula	2007	1657	3314	4706	0.4	0.2	1326	941	2267
Noss	2013	465	930	1321	0.4	0.2	372	264	636
Fair Isle	2013	266	532	755	0.4	0.2	213	151	364
Hoy	2010	1346	2692	3823	0.4	0.2	1077	765	1841
UK Non-SPA North Sea colonies	2000	3000	6000	8520	0.4	0.2	2400	1704	4104
Handa	2013	135	270	383	1	0.4	270	153	423
St Kilda	2012	181	362	514	1	0.4	362	206	568
UK Non-SPA western colonies	2000	100	200	284	1	0.4	200	114	314
Total overseas							2,532	889	3,421
Total UK							7,622	5,293	12,915
Total							10,154	6,182	16,336

Table 32. BDMPS for great skua in winter (November to February) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in winter	Proportion of immatures in UK North Sea & Channel waters in winter	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	1980s	5400	10800	15336	0.01	0.001	108	15	123
Norway	2010	360	720	1022	0.01	0.001	7	1	8
Faroe	2012	500	1000	1420	0.01	0.001	10	1	11
Hermaness, Saxavord	2013	979	1958	2780	0	0	0	0	0
Fetlar	2002	585	1170	1661	0	0	0	0	0
Ronas Hill, North Roe	2002	189	378	537	0	0	0	0	0
Foula	2007	1657	3314	4706	0	0	0	0	0
Noss	2013	465	930	1321	0	0	0	0	0
Fair Isle	2013	266	532	755	0	0	0	0	0
Hoy	2010	1346	2692	3823	0	0	0	0	0
UK Non-SPA North Sea colonies	2000	3000	6000	8520	0	0	0	0	0
Handa	2013	135	270	383	0	0	0	0	0
St Kilda	2012	181	362	514	0	0	0	0	0
UK Non-SPA western colonies	2000	100	200	284	0	0	0	0	0
Total overseas							125	18	143
Total UK							0	0	0
Total							125	18	143

Table 33. BDMPS for great skua in winter (November to February) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in winter	Proportion of immatures in UK western waters in winter	UK western Number adults	UK western Number immatures	UK western Total birds
Iceland	1980s	5400	10800	15336	0.1	0.001	1080	15	1095
Norway	2010	360	720	1022	0.1	0.001	72	1	73
Faroe	2012	500	1000	1420	0.05	0.001	50	1	51
Hermaness, Saxavord	2013	979	1958	2780	0.01	0	20	0	20
Fetlar	2002	585	1170	1661	0.01	0	12	0	12
Ronas Hill, North Roe	2002	189	378	537	0.01	0	4	0	4
Foula	2007	1657	3314	4706	0.01	0	33	0	33
Noss	2013	465	930	1321	0.01	0	9	0	9
Fair Isle	2013	266	532	755	0.01	0	5	0	5
Hoy	2010	1346	2692	3823	0.01	0	27	0	27
UK Non-SPA North Sea colonies	2000	3000	6000	8520	0.01	0	60	0	60
Handa	2013	135	270	383	0.01	0	3	0	3
St Kilda	2012	181	362	514	0.01	0	4	0	4
UK Non-SPA western colonies	2000	100	200	284	0.01	0	2	0	2
Total overseas							1,202	18	1,220
Total UK							178	0	178
Total							1,380	18	1,398

Table 34. BDMPS for great skua in spring migration (March-April) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in spring	Proportion of immatures in UK North Sea & Channel waters in spring	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	1980s	5400	10800	15336	0.05	0.02	540	307	847
Norway	2010	360	720	1022	0.05	0.02	36	20	56
Faroe	2012	500	1000	1420	0.05	0.02	50	28	78
Hermaness, Saxavord	2013	979	1958	2780	0.3	0.1	587	278	865
Fetlar	2002	585	1170	1661	0.3	0.1	351	166	517
Ronas Hill, North Roe	2002	189	378	537	0.3	0.1	113	54	167
Foula	2007	1657	3314	4706	0.3	0.1	994	471	1465
Noss	2013	465	930	1321	0.3	0.1	279	132	411
Fair Isle	2013	266	532	755	0.3	0.1	160	76	235
Hoy	2010	1346	2692	3823	0.3	0.1	808	382	1190
UK Non-SPA North Sea colonies	2000	3000	6000	8520	0.3	0.1	1800	852	2652
Handa	2013	135	270	383	0	0	0	0	0
St Kilda	2012	181	362	514	0	0	0	0	0
UK Non-SPA western colonies	2000	100	200	284	0	0	0	0	0
Total overseas							626	356	982
Total UK							5,092	2,410	7,503
Total							5,718	2,766	8,485

Table 35. BDMPS for great skua in spring migration (March-April) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in spring	Proportion of immatures in UK western waters in spring	UK western Number adults	UK western Number immatures	UK western Total birds
Iceland	1980s	5400	10800	15336	0.3	0.05	3240	767	4007
Norway	2010	360	720	1022	0.2	0.05	144	51	195
Faroe	2012	500	1000	1420	0.4	0.05	400	71	471
Hermaness, Saxavord	2013	979	1958	2780	0.7	0.3	1371	834	2205
Fetlar	2002	585	1170	1661	0.7	0.3	819	498	1317
Ronas Hill, North Roe	2002	189	378	537	0.7	0.3	265	161	426
Foula	2007	1657	3314	4706	0.7	0.3	2320	1412	3732
Noss	2013	465	930	1321	0.7	0.3	651	396	1047
Fair Isle	2013	266	532	755	0.7	0.3	372	227	599
Hoy	2010	1346	2692	3823	0.7	0.3	1884	1147	3031
UK Non-SPA North Sea colonies	2000	3000	6000	8520	0.7	0.3	4200	2556	6756
Handa	2013	135	270	383	1	0.4	270	153	423
St Kilda	2012	181	362	514	1	0.4	362	206	568
UK Non-SPA western colonies	2000	100	200	284	1	0.4	200	114	314
Total overseas							3,784	889	4,673
Total UK							12,714	7,704	20,417
Total							16,498	8,593	25,090

Table 36. BDMPS for lesser black-backed gull in autumn migration (August-October) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in autumn	Proportion of immatures in UK North Sea & Channel waters in autumn	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	1990s	25000	50000	34000	0.2	0.1	10000	3400	13400
Norway	1990s	30000	60000	40800	0.3	0.1	18000	4080	22080
Faroe	2012	9000	18000	12240	0.4	0.2	7200	2448	9648
Sweden	1990s	18000	36000	24480	0.1	0.05	3600	1224	4824
Denmark	1990s	4400	8800	5984	0.1	0.05	880	299	1179
Ireland	2000	3800	7600	5168	0.1	0.05	760	258	1018
Netherlands	2012	80000	160000	108800	0.05	0.025	8000	2720	10720
Forth Islands	2005-09	1608	3216	2187	1	0.7	3216	1531	4747
Alde-Ore Estuary	2012	640	1280	870	1	0.7	1280	609	1889
UK North Sea non-SPA cols	2000	13000	26000	17680	1	0.7	26000	12376	38376
Ailsa Craig	2010	183	366	249	0.5	0.4	183	100	283
Rathlin Island	2011	107	214	146	0.5	0.4	107	58	165
Lough Neagh & Lough Beg	2000	493	986	670	0.5	0.4	493	268	761
Bowland Fells	2008-12	4575	9150	6222	0.5	0.4	4575	2489	7064
Morcambe Bay	2012	4987	9974	6782	0.5	0.4	4987	2713	7700
Ribble & Alt Estuaries	2012	8267	16534	11243	0.5	0.4	8267	4497	12764
Skokholm, Skomer, Mholm	2013	9640	19280	13110	0.3	0.3	5784	3933	9717
Isles of Scilly	2006	3400	6800	4624	0.1	0.05	680	231	911
UK Western non-SPA colonies	2000	40000	80000	54400	0.5	0.4	40000	21760	61760
Total overseas							48,440	14,430	62,870
Total UK							95,572	50,565	146,137
Total							144,012	64,995	209,007

Table 37. BDMPS for lesser black-backed gull in autumn migration (August-October) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in autumn	Proportion immatures in UK western waters in autumn	UK western Number adults	UK western Number immatures	UK western Total birds
Iceland	1990s	25000	50000	34000	0.2	0.1	10000	3400	13400
Norway	1990s	30000	60000	40800	0.1	0.05	6000	2040	8040
Faroe	2012	9000	18000	12240	0.4	0.2	7200	2448	9648
Sweden	1990s	18000	36000	24480	0.05	0.02	1800	490	2290
Denmark	1990s	4400	8800	5984	0.05	0.02	440	120	560
Ireland	2000	3800	7600	5168	0.4	0.2	3040	1034	4074
Netherlands	2012	80000	160000	108800	0.025	0.01	4000	1088	5088
Forth Islands	2005-09	1608	3216	2187	0	0.1	0	219	219
Alde-Ore Estuary	2012	640	1280	870	0	0.1	0	87	87
UK North Sea non-SPA cols	2000	13000	26000	17680	0	0.1	0	1768	1768
Ailsa Craig	2010	183	366	249	0.5	0.4	183	100	283
Rathlin Island	2011	107	214	146	0.5	0.4	107	58	165
Lough Neagh & Lough Beg	2000	493	986	670	0.5	0.4	493	268	761
Bowland Fells	2008-12	4575	9150	6222	0.5	0.4	4575	2489	7064
Morcambe Bay	2012	4987	9974	6782	0.5	0.4	4987	2713	7700
Ribble & Alt Estuaries	2012	8267	16534	11243	0.5	0.4	8267	4497	12764
Skokholm, Skomer, Mholm	2013	9640	19280	13110	0.7	0.4	13496	5244	18740
Isles of Scilly	2006	3400	6800	4624	0.9	0.6	6120	2774	8894
UK Western non-SPA cols	2000	40000	80000	54400	0.5	0.4	40000	21760	61760
Total overseas							32,480	10,619	43,099
Total UK							78,228	41,977	120,205
Total							110,708	52,596	163,304

Table 38. BDMPS for lesser black-backed gull in winter (November to February) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in winter	Proportion immatures in UK North Sea & Channel waters in winter	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	1990s	25000	50000	34000	0.05	0	2500	0	2500
Norway	1990s	30000	60000	40800	0.05	0	3000	0	3000
Faroe	2012	9000	18000	12240	0.05	0	900	0	900
Sweden	1990s	18000	36000	24480	0.01	0	360	0	360
Denmark	1990s	4400	8800	5984	0.01	0	88	0	88
Ireland	2000	3800	7600	5168	0.01	0	76	0	76
Netherlands	2012	80000	160000	108800	0.005	0	800	0	800
Forth Islands	2005-09	1608	3216	2187	0.5	0.05	1608	109	1717
Alde-Ore Estuary	2012	640	1280	870	0.5	0.05	640	44	684
UK North Sea non-SPA cols	2000	13000	26000	17680	0.5	0.05	13000	884	13884
Ailsa Craig	2010	183	366	249	0.1	0.01	37	2	39
Rathlin Island	2011	107	214	146	0.1	0.01	21	1	23
Lough Neagh & Lough Beg	2000	493	986	670	0.1	0.01	99	7	105
Bowland Fells	2008-12	4575	9150	6222	0.1	0.01	915	62	977
Morcambe Bay	2012	4987	9974	6782	0.1	0.01	997	68	1065
Ribble & Alt Estuaries	2012	8267	16534	11243	0.1	0.01	1653	112	1766
Skokholm, Skomer, Mholm	2013	9640	19280	13110	0.1	0.01	1928	131	2059
Isles of Scilly	2006	3400	6800	4624	0.1	0.01	680	46	726
UK Western non-SPA cols	2000	40000	80000	54400	0.1	0.01	8000	544	8544
Total overseas							7,724	0	7,724
Total UK							29,578	2,011	31,590
Total							37,302	2,011	39,314

Table 39. BDMPS for lesser black-backed gull in winter (November to February) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in winter	Proportion immatures in UK western waters in winter	UK western Number adults	UK western Number immatures	UK western Total birds
Iceland	1990s	25000	50000	34000	0.05	0	2500	0	2500
Norway	1990s	30000	60000	40800	0.02	0	1200	0	1200
Faroe	2012	9000	18000	12240	0.05	0	900	0	900
Sweden	1990s	18000	36000	24480	0.01	0	360	0	360
Denmark	1990s	4400	8800	5984	0.01	0	88	0	88
Ireland	2000	3800	7600	5168	0.2	0.05	1520	258	1778
Netherlands	2012	80000	160000	108800	0.005	0	800	0	800
Forth Islands	2005-09	1608	3216	2187	0	0	0	0	0
Alde-Ore Estuary	2012	640	1280	870	0	0	0	0	0
UK North Sea non-SPA cols	2000	13000	26000	17680	0	0	0	0	0
Ailsa Craig	2010	183	366	249	0.2	0.05	73	12	86
Rathlin Island	2011	107	214	146	0.2	0.05	43	7	50
Lough Neagh & Lough Beg	2000	493	986	670	0.2	0.05	197	34	231
Bowland Fells	2008-12	4575	9150	6222	0.2	0.05	1830	311	2141
Morcambe Bay	2012	4987	9974	6782	0.2	0.05	1995	339	2334
Ribble & Alt Estuaries	2012	8267	16534	11243	0.2	0.05	3307	562	3869
Skokholm, Skomer, Mholm	2013	9640	19280	13110	0.2	0.05	3856	656	4511
Isles of Scilly	2006	3400	6800	4624	0.2	0.05	1360	231	1591
UK Western non-SPA cols	2000	40000	80000	54400	0.2	0.05	16000	2720	18720
Total overseas							7,368	258	7,626
Total UK							28,661	4,872	33,533
Total							36,029	5,130	41,159

Table 40. BDMPS for lesser black-backed gull in spring migration (March-April) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in spring	Proportion immatures in UK North Sea & Channel waters in spring	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Iceland	1990s	25000	50000	34000	0.1	0.05	5000	1700	6700
Norway	1990s	30000	60000	40800	0.3	0.1	18000	4080	22080
Faroe	2012	9000	18000	12240	0.2	0.1	3600	1224	4824
Sweden	1990s	18000	36000	24480	0.1	0.05	3600	1224	4824
Denmark	1990s	4400	8800	5984	0.1	0.05	880	299	1179
Ireland	2000	3800	7600	5168	0.1	0.05	760	258	1018
Netherlands	2012	80000	160000	108800	0.05	0.025	8000	2720	10720
Forth Islands	2005-09	1608	3216	2187	1	0.7	3216	1531	4747
Alde-Ore Estuary	2012	640	1280	870	1	0.7	1280	609	1889
UK North Sea non-SPA cols	2000	13000	26000	17680	1	0.7	26000	12376	38376
Ailsa Craig	2010	183	366	249	0.5	0.4	183	100	283
Rathlin Island	2011	107	214	146	0.5	0.4	107	58	165
Lough Neagh & Lough Beg	2000	493	986	670	0.5	0.4	493	268	761
Bowland Fells	2008-12	4575	9150	6222	0.5	0.4	4575	2489	7064
Morcambe Bay	2012	4987	9974	6782	0.5	0.4	4987	2713	7700
Ribble & Alt Estuaries	2012	8267	16534	11243	0.5	0.4	8267	4497	12764
Skokholm, Skomer, Mholm	2013	9640	19280	13110	0.3	0.3	5784	3933	9717
Isles of Scilly	2006	3400	6800	4624	0.1	0.05	680	231	911
UK Western non-SPA cols	2000	40000	80000	54400	0.5	0.4	40000	21760	61760
Total overseas							39,840	11,506	51,346
Total UK							95,572	50,565	146,137
Total							135,412	62,071	197,483

Table 41. BDMPS for lesser black-backed gull in spring migration (March-April) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in spring	Proportion immatures in UK western waters in spring	UK western Number adults	UK western Number immatures	UK western Total birds
Iceland	1990s	25000	50000	34000	0.2	0.1	10000	3400	13400
Norway	1990s	30000	60000	40800	0.1	0.05	6000	2040	8040
Faroe	2012	9000	18000	12240	0.4	0.2	7200	2448	9648
Sweden	1990s	18000	36000	24480	0.05	0.02	1800	490	2290
Denmark	1990s	4400	8800	5984	0.05	0.02	440	120	560
Ireland	2000	3800	7600	5168	0.4	0.2	3040	1034	4074
Netherlands	2012	80000	160000	108800	0.025	0.01	4000	1088	5088
Forth Islands	2005-09	1608	3216	2187	0	0.1	0	219	219
Alde-Ore Estuary	2012	640	1280	870	0	0.1	0	87	87
UK North Sea non-SPA	2000	13000	26000	17680	0	0.1	0	1768	1768
Ailsa Craig	2010	183	366	249	0.5	0.4	183	100	283
Rathlin Island	2011	107	214	146	0.5	0.4	107	58	165
Lough Neagh & L. Beg	2000	493	986	670	0.5	0.4	493	268	761
Bowland Fells	2008-12	4575	9150	6222	0.5	0.4	4575	2489	7064
Morcambe Bay	2012	4987	9974	6782	0.5	0.4	4987	2713	7700
Ribble & Alt Estuaries	2012	8267	16534	11243	0.5	0.4	8267	4497	12764
Skokholm, Skomer, Mholm	2013	9640	19280	13110	0.7	0.4	13496	5244	18740
Isles of Scilly	2006	3400	6800	4624	0.9	0.6	6120	2774	8894
UK Western non-SPA cols	2000	40000	80000	54400	0.5	0.4	40000	21760	61760
Total overseas							32,480	10,619	43,100
Total UK							78,228	41,977	120,205
Total							110,708	52,596	163,305

Table 42. BDMPS for herring gull in non-breeding season (September to February) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters non-breeding season	Proportion immatures in UK North Sea & Channel waters non-breeding season	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Barents Sea	2000	126000	252000	274680	0.2	0.3	50400	82404	132804
Faroe	2012	1500	3000	3270	0.2	0.3	600	981	1581
Ireland	2000	5000	10000	10900	0.02	0.05	200	545	745
East Caithness Cliffs	1999	3393	6786	7397	0.99	0.95	6718	7027	13745
Troup, Pennan & Lions	2007	1597	3194	3481	0.99	0.95	3162	3307	6469
Buchan Ness to Collieston	2010	3114	6228	6789	0.99	0.95	6166	6449	12615
Fowlsheugh	2012	259	518	565	0.99	0.95	513	536	1049
Forth Islands	2005-09	2827	5654	6163	0.99	0.95	5597	5855	11452
St Abbs Head/ Fast Castle	2013	239	478	521	0.99	0.95	473	495	968
Flamborough & Filey Coast	2010	495	990	1079	0.99	0.95	980	1025	2005
Alde-Ore Estuary	2006	800	1600	1744	0.99	0.95	1584	1657	3241
UK North Sea non-SPA cols	2000	65000	130000	141700	0.99	0.95	128700	134615	263315
Canna & Sanday	2011	63	126	137	0.05	0.1	6	14	20
Ailsa Craig	2013	129	258	281	0.05	0.1	13	28	41
Rathlin Island	2011	28	56	61	0.05	0.1	3	6	9
Morecambe Bay	2012	1734	3468	3780	0.05	0.1	173	378	551
UK western non-SPA cols	2000	50000	100000	109000	0.05	0.1	5000	10900	15900
Total overseas							51,200	83,930	135,130
Total UK							159,089	172,292	331,381
Total							210,289	256,222	466,511

Table 43. BDMPS for herring gull in non-breeding season (September to February) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters non-breeding season	Proportion immatures in UK western waters non-breeding season	UK western Number adults	UK western Number immatures	UK western Total birds
Barents Sea	2000	126000	252000	274680	0.001	0.005	252	1373	1625
Faroe	2012	1500	3000	3270	0.2	0.3	600	981	1581
Ireland	2000	5000	10000	10900	0.3	0.4	3000	4360	7360
East Caithness Cliffs	1999	3393	6786	7397	0.001	0.001	7	7	14
Troup, Pennan & Lions	2007	1597	3194	3481	0.001	0.001	3	3	7
Buchan Ness to Collieston	2010	3114	6228	6789	0.001	0.001	6	7	13
Fowlsheugh	2012	259	518	565	0.001	0.001	1	1	1
Forth Islands	2005-09	2827	5654	6163	0.001	0.001	6	6	12
St Abbs Head/ Fast Castle	2013	239	478	521	0.001	0.001	0	1	1
Flamborough & Filey Coast	2010	495	990	1079	0.001	0.001	1	1	2
Alde-Ore Estuary	2006	800	1600	1744	0.001	0.001	2	2	3
UK North Sea non-SPA cols	2000	65000	130000	141700	0.001	0.001	130	142	272
Canna & Sanday	2011	63	126	137	0.8	0.7	101	96	197
Ailsa Craig	2013	129	258	281	0.8	0.7	206	197	403
Rathlin Island	2011	28	56	61	0.8	0.7	45	43	88
Morecambe Bay	2012	1734	3468	3780	0.8	0.7	2774	2646	5420
UK western non-SPA cols	2000	50000	100000	109000	0.8	0.7	80000	76300	156300
Total overseas							3,852	6,714	10,566
Total UK							83,282	79,451	162,733
Total							87,134	86,165	173,299

Table 44. BDMPS for great black-backed gull in non-breeding season (September to March) in 'UK North Sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea waters non-breeding season	Proportion of immatures in UK North Sea waters non-breeding season	UK N Sea Number adults	UK N Sea Number immatures	UK N Sea Total birds
Barents Sea	1990s*	33000	66000	83160	0.3	0.5	19800	41580	61380
Faroe	2012	1000	2000	2520	0.3	0.3	600	756	1356
Ireland	2000	2000	4000	5040	0	0	0	0	0
Calf of Eday	2006	281	562	708	1	1	562	708	1270
Copinsay	2010	218	436	549	1	1	436	549	985
Hoy	2011	60	120	151	1	1	120	151	271
East Caithness Cliffs	1999	175	350	441	1	1	350	441	791
UK North Sea non-SPA colonies	2000	5000	10000	12600	1	1	10000	12600	22600
North Rona & Sula Sgeir	2012	191	382	481	0.01	0.1	4	48	52
Isles of Scilly	2006	901	1802	2271	0.01	0.1	18	227	245
UK western non-SPA colonies	2000	9000	18000	22680	0.01	0.1	180	2268	2448
Total overseas							20,400	42,336	62,736
Total UK							11,670	16,993	28,663
Total							32,070	59,329	91,399

*updated to 2012 by R.T. Barrett pers. comm.

Table 45. BDMPS for great black-backed gull in non-breeding season (September to March) in 'UK west of Scotland waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in west of Scotland waters non-breeding season	Proportion of immatures in UK west of Scotland waters non-breeding season	west of Scotland Number adults	west of Scotland Number immatures	west of Scotland Total birds
Barents Sea	1990s*	33000	66000	83160	0.01	0.08	660	6653	7313
Faroe	2012	1000	2000	2520	0.1	0.3	200	756	956
Ireland	2000	2000	4000	5040	0.1	0.2	400	1008	1408
Calf of Eday	2006	281	562	708	0	0	0	0	0
Copinsay	2010	218	436	549	0	0	0	0	0
Hoy	2011	60	120	151	0	0	0	0	0
East Caithness Cliffs	1999	175	350	441	0	0	0	0	0
UK North Sea non-SPA colonies	2000	5000	10000	12600	0	0	0	0	0
North Rona & Sula Sgeir	2012	191	382	481	0.99	0.8	378	385	763
Isles of Scilly	2006	901	1802	2271	0	0	0	0	0
UK western non-SPA colonies	2000	9000	18000	22680	0.7	0.5	12600	11340	23940
Total overseas							1,260	8,417	9,677
Total UK							12,978	11,725	24,703
Total							14,238	20,142	34,380

*updated to 2012 by R.T. Barrett pers. comm.

Table 46. BDMPS for great black-backed gull in non-breeding season (September to March) in 'UK south-west & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK south-west & Channel waters non-breeding season	Proportion of immatures in UK south-west & Channel waters non-breeding season	UK south-west & Channel Number adults	UK south-west & Channel Number immatures	UK south-west & Channel Total birds
Barents Sea	1990s*	33000	66000	83160	0	0.02	0	1663	1663
Faroe	2012	1000	2000	2520	0	0.2	0	504	504
Ireland	2000	2000	4000	5040	0.1	0.3	400	1512	1912
Calf of Eday	2006	281	562	708	0	0	0	0	0
Copinsay	2010	218	436	549	0	0	0	0	0
Hoy	2011	60	120	151	0	0	0	0	0
East Caithness Cliffs	1999	175	350	441	0	0	0	0	0
UK North Sea non-SPA colonies	2000	5000	10000	12600	0	0	0	0	0
North Rona & Sula Sgeir	2012	191	382	481	0	0.1	0	48	48
Isles of Scilly	2006	901	1802	2271	0.9	0.7	1622	1589	3211
UK western non-SPA colonies	2000	9000	18000	22680	0.2	0.3	3600	6804	10404
Total overseas							400	3,679	4,079
Total UK							5,222	8,441	13,663
Total							5,622	12,120	17,742

*updated to 2012 by R.T. Barrett pers. comm.

Table 47. BDMPS for black-legged kittiwake in autumn migration (August to December) in 'UK North Sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea waters in autumn	Proportion of immatures in UK North Sea waters in autumn	UK N Sea Number adults	UK N Sea Number immatures	UK N Sea Total birds
Russia	2000	140000	280000	246400	0.1	0.1	28000	24640	52640
Norway	2010	700000	1400000	1232000	0.1	0.1	140000	123200	263200
Faroe	2012	200000	400000	352000	0.1	0.1	40000	35200	75200
Germany	2010	6000	12000	10560	0.1	0.1	1200	1056	2256
France	2010	4000	8000	7040	0.05	0.05	400	352	752
Ireland	2000	20000	40000	35200	0.05	0.05	2000	1760	3760
Hermaness, Saxavord	2009	391	782	688	0.6	0.4	469	275	744
Foula	2013	327	654	576	0.6	0.4	392	230	623
Noss	2010	507	1014	892	0.6	0.4	608	357	965
Sumburgh Head	2013	210	420	370	0.6	0.4	252	148	400
Fair Isle	2013	771	1542	1357	0.6	0.4	925	543	1468
West Westray	2007	12055	24110	21217	0.6	0.4	14466	8487	22953
Calf of Eday	2006	747	1494	1315	0.6	0.4	896	526	1422
Marwick Head	2013	526	1052	926	0.6	0.4	631	370	1002
Rousay	2009	1764	3528	3105	0.6	0.4	2117	1242	3359
Copinsay	2012	666	1332	1172	0.6	0.4	799	469	1268
Hoy	2007	397	794	699	0.6	0.4	476	279	756
North Caithness Cliffs	2000	10150	20300	17864	0.6	0.4	12180	7146	19326
East Caithness Cliffs	1999	40410	80820	71122	0.6	0.4	48492	28449	76941
Troup, Pennan & Lions Heads	2007	14896	29792	26217	0.6	0.4	17875	10487	28362
Buchan Ness to Collieston	2007	12542	25084	22074	0.6	0.4	15050	8830	23880
Fowlsheugh	2012	9337	18674	16433	0.6	0.4	11204	6573	17778
Forth Islands	2013	3100	6200	5456	0.6	0.4	3720	2182	5902
St Abbs Head to Fast Castle	2013	3403	6806	5989	0.6	0.4	4084	2396	6479
Farne Islands	2013	3443	6886	6060	0.6	0.4	4132	2424	6555

Flamborough and Filey	2008	37617	75234	66206	0.6	0.4	45140	26482	71623
UK North Sea non-SPA colonies	2000*	70000	140000	123200	0.6	0.4	84000	49280	133280
Cape Wrath	2000	10344	20688	18205	0.01	0.05	207	910	1117
North Rona & Sula Sgeir	2012	1253	2506	2205	0.01	0.05	25	110	135
Handa	2013	1872	3744	3295	0.01	0.05	37	165	202
St Kilda	2008	957	1914	1684	0.01	0.05	19	84	103
Flannan Isles	1998	1392	2784	2450	0.01	0.05	28	122	150
Shiant Isles	2008	549	1098	966	0.01	0.05	11	48	59
Canna & Sanday	2013	820	1640	1443	0.01	0.05	16	72	89
Rum	2000	788	1576	1387	0.01	0.05	16	69	85
Mingulay & Berneray	2009	2228	4456	3921	0.01	0.05	45	196	241
North Colonsay & Western Cliffs	2000	5563	11126	9791	0.01	0.05	111	490	601
Ailsa Craig	2013	489	978	861	0.01	0.05	10	43	53
Rathlin Island	2011	7922	15844	13943	0.01	0.05	158	697	856
Skomer, Skokholm, Middleholm	2013	1045	2090	1839	0.01	0.05	21	92	113
UK Western non-SPA colonies	2000*	30000	60000	52800	0.01	0.05	600	2640	3240
Total overseas							211,600	186,208	397,808
Total UK							269,215	162,914	432,129
Total							480,815	349,122	829,937

*updated to 2012 using trend in Foster & Marrs 2012 and JNCC database

Table 48. BDMPS for black-legged kittiwake in autumn migration (August to December) in 'UK western waters & Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters (including Channel) in autumn	Proportion of immatures in UK western waters (including Channel) waters in autumn	UK western waters & Channel Number adults	UK western waters & Channel Number immatures	UK western waters & Channel Total birds
Russia	2000	140000	280000	246400	0.1	0.1	28000	24640	52640
Norway	2010	700000	1400000	1232000	0.15	0.15	210000	184800	394800
Faroe	2012	200000	400000	352000	0.2	0.2	80000	70400	150400
Germany	2010	6000	12000	10560	0.05	0.05	600	528	1128
France	2010	4000	8000	7040	0.1	0.1	800	704	1504
Ireland	2000	20000	40000	35200	0.3	0.2	12000	7040	19040
Hermaness, Saxavord	2009	391	782	688	0.2	0.2	156	138	294
Foula	2013	327	654	576	0.2	0.2	131	115	246
Noss	2010	507	1014	892	0.2	0.2	203	178	381
Sumburgh Head	2013	210	420	370	0.2	0.2	84	74	158
Fair Isle	2013	771	1542	1357	0.2	0.2	308	271	580
West Westray	2007	12055	24110	21217	0.2	0.2	4822	4243	9065
Calf of Eday	2006	747	1494	1315	0.2	0.2	299	263	562
Marwick Head	2013	526	1052	926	0.2	0.2	210	185	396
Rousay	2009	1764	3528	3105	0.2	0.2	706	621	1327
Copinsay	2012	666	1332	1172	0.2	0.2	266	234	501
Hoy	2007	397	794	699	0.2	0.2	159	140	299
North Caithness Cliffs	2000	10150	20300	17864	0.2	0.2	4060	3573	7633
East Caithness Cliffs	1999	40410	80820	71122	0.2	0.2	16164	14224	30388
Troup, Pennan & Lions Heads	2007	14896	29792	26217	0.2	0.2	5958	5243	11202
Buchan Ness to Collieston	2007	12542	25084	22074	0.2	0.2	5017	4415	9432
Fowlsheugh	2012	9337	18674	16433	0.2	0.2	3735	3287	7021
Forth Islands	2013	3100	6200	5456	0.2	0.2	1240	1091	2331

St Abbs Head to Fast Castle	2013	3403	6806	5989	0.2	0.2	1361	1198	2559
Farne Islands	2013	3443	6886	6060	0.2	0.2	1377	1212	2589
Flamborough and Filey	2008	37617	75234	66206	0.2	0.2	15047	13241	28288
UK North Sea non-SPA colonies	2000*	70000	140000	123200	0.2	0.2	28000	24640	52640
Cape Wrath	2000	10344	20688	18205	0.6	0.4	12413	7282	19695
North Rona & Sula Sgeir	2012	1253	2506	2205	0.6	0.4	1504	882	2386
Handa	2013	1872	3744	3295	0.6	0.4	2246	1318	3564
St Kilda	2008	957	1914	1684	0.6	0.4	1148	674	1822
Flannan Isles	1998	1392	2784	2450	0.6	0.4	1670	980	2650
Shiant Isles	2008	549	1098	966	0.6	0.4	659	386	1045
Canna & Sanday	2013	820	1640	1443	0.6	0.4	984	577	1561
Rum	2000	788	1576	1387	0.6	0.4	946	555	1500
Mingulay & Berneray	2009	2228	4456	3921	0.6	0.4	2674	1569	4242
North Colonsay & Western Cliffs	2000	5563	11126	9791	0.6	0.4	6676	3916	10592
Ailsa Craig	2013	489	978	861	0.6	0.4	587	344	931
Rathlin Island	2011	7922	15844	13943	0.6	0.4	9506	5577	15083
Skomer, Skokholm, Middleholm	2013	1045	2090	1839	0.6	0.4	1254	736	1990
UK Western non-SPA colonies	2000*	30000	60000	52800	0.6	0.4	36000	21120	57120
Total overseas							331,400	288,112	619,512
Total UK							167,570	124,503	292,074
Total							498,970	412,615	911,586

*updated to 2012 using trend in Foster & Marrs 2012 and JNCC database

Table 49. BDMPS for black-legged kittiwake in spring migration (January to April) in 'UK North Sea waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea waters in spring	Proportion of immatures in UK North Sea waters in spring	UK N Sea Number adults	UK N Sea Number immatures	UK N Sea Total birds
Russia	2000	140000	280000	246400	0.05	0.07	14000	17248	31248
Norway	2010	700000	1400000	1232000	0.05	0.07	70000	86240	156240
Faroe	2012	200000	400000	352000	0.05	0.07	20000	24640	44640
Germany	2010	6000	12000	10560	0.15	0.25	1800	2640	4440
France	2010	4000	8000	7040	0.05	0.1	400	704	1104
Ireland	2000	20000	40000	35200	0.01	0.01	400	352	752
Hermaness, Saxavord	2009	391	782	688	0.6	0.3	469	206	676
Foula	2013	327	654	576	0.6	0.3	392	173	565
Noss	2010	507	1014	892	0.6	0.3	608	268	876
Sumburgh Head	2013	210	420	370	0.6	0.3	252	111	363
Fair Isle	2013	771	1542	1357	0.6	0.3	925	407	1332
West Westray	2007	12055	24110	21217	0.6	0.3	14466	6365	20831
Calf of Eday	2006	747	1494	1315	0.6	0.3	896	394	1291
Marwick Head	2013	526	1052	926	0.6	0.3	631	278	909
Rousay	2009	1764	3528	3105	0.6	0.3	2117	931	3048
Copinsay	2012	666	1332	1172	0.6	0.3	799	352	1151
Hoy	2007	397	794	699	0.6	0.3	476	210	686
North Caithness Cliffs	2000	10150	20300	17864	0.6	0.3	12180	5359	17539
East Caithness Cliffs	1999	40410	80820	71122	0.6	0.3	48492	21336	69828
Troup, Pennan & Lions Heads	2007	14896	29792	26217	0.6	0.3	17875	7865	25740
Buchan Ness to Collieston	2007	12542	25084	22074	0.6	0.3	15050	6622	21673
Fowlsheugh	2012	9337	18674	16433	0.6	0.3	11204	4930	16134
Forth Islands	2013	3100	6200	5456	0.6	0.3	3720	1637	5357
St Abbs Head to Fast Castle	2013	3403	6806	5989	0.6	0.3	4084	1797	5880
Farne Islands	2013	3443	6886	6060	0.6	0.3	4132	1818	5950

Flamborough and Filey	2008	37617	75234	66206	0.6	0.3	45140	19862	65002
UK North Sea non-SPA colonies	2000*	70000	140000	123200	0.6	0.3	84000	36960	120960
Cape Wrath	2000	10344	20688	18205	0.01	0.02	207	364	571
North Rona & Sula Sgeir	2012	1253	2506	2205	0.01	0.02	25	44	69
Handa	2013	1872	3744	3295	0.01	0.02	37	66	103
St Kilda	2008	957	1914	1684	0.01	0.02	19	34	53
Flannan Isles	1998	1392	2784	2450	0.01	0.02	28	49	77
Shiant Isles	2008	549	1098	966	0.01	0.02	11	19	30
Canna & Sanday	2013	820	1640	1443	0.01	0.02	16	29	45
Rum	2000	788	1576	1387	0.01	0.02	16	28	43
Mingulay & Berneray	2009	2228	4456	3921	0.01	0.02	45	78	123
North Colonsay & Western Cliffs	2000	5563	11126	9791	0.01	0.02	111	196	307
Ailsa Craig	2013	489	978	861	0.01	0.02	10	17	27
Rathlin Island	2011	7922	15844	13943	0.01	0.02	158	279	437
Skomer, Skokholm, Middleholm	2013	1045	2090	1839	0.01	0.02	21	37	58
UK Western non-SPA colonies	2000*	30000	60000	52800	0.01	0.02	600	1056	1656
Total overseas							106,600	131,824	238,424
Total UK							269,215	120,177	389,392
Total							375,815	252,001	627,816

*updated to 2012 using trend in Foster & Marrs 2012 and JNCC database

Table 50. BDMPS for black-legged kittiwake in spring migration (January to April) in 'UK western waters & Channel'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters (including Channel) in spring	Proportion of immatures in UK western waters (including Channel) waters in spring	UK western waters & Channel Number adults	UK western waters & Channel Number immatures	UK western waters & Channel Total birds
Russia	2000	140000	280000	246400	0.05	0.1	14000	24640	38640
Norway	2010	700000	1400000	1232000	0.05	0.1	70000	123200	193200
Faroe	2012	200000	400000	352000	0.1	0.1	40000	35200	75200
Germany	2010	6000	12000	10560	0.05	0.05	600	528	1128
France	2010	4000	8000	7040	0.1	0.1	800	704	1504
Ireland	2000	20000	40000	35200	0.3	0.2	12000	7040	19040
Hermaness, Saxavord	2009	391	782	688	0.3	0.2	235	138	372
Foula	2013	327	654	576	0.3	0.2	196	115	311
Noss	2010	507	1014	892	0.3	0.2	304	178	483
Sumburgh Head	2013	210	420	370	0.3	0.2	126	74	200
Fair Isle	2013	771	1542	1357	0.3	0.2	463	271	734
West Westray	2007	12055	24110	21217	0.3	0.2	7233	4243	11476
Calf of Eday	2006	747	1494	1315	0.3	0.2	448	263	711
Marwick Head	2013	526	1052	926	0.3	0.2	316	185	501
Rousay	2009	1764	3528	3105	0.3	0.2	1058	621	1679
Copinsay	2012	666	1332	1172	0.3	0.2	400	234	634
Hoy	2007	397	794	699	0.3	0.2	238	140	378
North Caithness Cliffs	2000	10150	20300	17864	0.3	0.2	6090	3573	9663
East Caithness Cliffs	1999	40410	80820	71122	0.3	0.2	24246	14224	38470
Troup, Pennan & Lions Heads	2007	14896	29792	26217	0.3	0.2	8938	5243	14181
Buchan Ness to Collieston	2007	12542	25084	22074	0.3	0.2	7525	4415	11940
Fowlsheugh	2012	9337	18674	16433	0.3	0.2	5602	3287	8889
Forth Islands	2013	3100	6200	5456	0.3	0.2	1860	1091	2951

St Abbs Head to Fast Castle	2013	3403	6806	5989	0.3	0.2	2042	1198	3240
Farne Islands	2013	3443	6886	6060	0.3	0.2	2066	1212	3278
Flamborough and Filey	2008	37617	75234	66206	0.3	0.2	22570	13241	35811
UK North Sea non-SPA colonies	2000*	70000	140000	123200	0.3	0.2	42000	24640	66640
Cape Wrath	2000	10344	20688	18205	0.8	0.4	16550	7282	23833
North Rona & Sula Sgeir	2012	1253	2506	2205	0.8	0.4	2005	882	2887
Handa	2013	1872	3744	3295	0.8	0.4	2995	1318	4313
St Kilda	2008	957	1914	1684	0.8	0.4	1531	674	2205
Flannan Isles	1998	1392	2784	2450	0.8	0.4	2227	980	3207
Shiant Isles	2008	549	1098	966	0.8	0.4	878	386	1265
Canna & Sanday	2013	820	1640	1443	0.8	0.4	1312	577	1889
Rum	2000	788	1576	1387	0.8	0.4	1261	555	1816
Mingulay & Berneray	2009	2228	4456	3921	0.8	0.4	3565	1569	5133
North Colonsay & Western Cliffs	2000	5563	11126	9791	0.8	0.4	8901	3916	12817
Ailsa Craig	2013	489	978	861	0.8	0.4	782	344	1127
Rathlin Island	2011	7922	15844	13943	0.8	0.4	12675	5577	18252
Skomer, Skokholm, Middleholm	2013	1045	2090	1839	0.8	0.4	1672	736	2408
UK Western non-SPA colonies	2000*	30000	60000	52800	0.8	0.4	48000	21120	69120
Total overseas							137,400	191,312	328,712
Total UK							238,311	124,503	362,814
Total							375,711	315,815	691,526

*updated to 2012 using trend in Foster & Marrs 2012 and JNCC database

Table 51. BDMPS for Sandwich tern in migration seasons (July-September and March-May) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters migration	Proportion of immatures in UK North Sea & Channel waters on migration	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Norway & Sweden	1990s	700	1400	882	0.1	0.1	140	88	228
Denmark	1990s	4500	9000	5670	0.1	0.1	900	567	1467
Germany	1990s	9700	19400	12222	0.1	0.1	1940	1222	3162
Netherlands	1990s	14500	29000	18270	0.1	0.1	2900	1827	4727
Belgium	2000	1550	3100	1953	0.1	0.1	310	195	505
Ireland	2000	1800	3600	2268	0	0	0	0	0
Loch of Strathbeg	2013	0	0	0	1	0.7	0	0	0
Ythan Estuary	2013	565	1130	712	1	0.7	1130	498	1628
Forth Islands	2013	0	0	0	1	0.7	0	0	0
Farne Islands	2013	824	1648	1038	1	0.7	1648	727	2375
Coquet Island	2013	670	1340	844	1	0.7	1340	591	1931
North Norfolk Coast	2012	4135	8270	5210	1	0.7	8270	3647	11917
Alde-Ore Estuary	2009	2	4	3	1	0.7	4	2	6
Foulness	2006	0	0	0	1	0.7	0	0	0
Chichester & Langstone Harb	2013	6	12	8	1	0.7	12	5	17
Solent & Southampton Water	2008	0	0	0	1	0.7	0	0	0
UK North Sea non-SPA colonies	2000	3500	7000	4410	1	0.7	7000	3087	10087
Carlingford Lough	2013	0	0	0	0	0	0	0	0
Larne Lough	2013	257	514	324	0	0	0	0	0
Strangford Lough	2012	771	1542	971	0	0	0	0	0
Morecambe Bay	2011	1	2	1	0	0	0	0	0
Duddon Estuary	2012	1	2	1	0	0	0	0	0
Ynys Feurig, Cemlyn Bay	2009	0	0	0	0	0	0	0	0
UK Western non-SPA colonies	2000	1500	3000	1890	0	0	0	0	0

Total overseas	6,190	3,900	10,090
Total UK	19,404	8,557	27,961
Total	25,594	12,457	38,051

Table 52. BDMPS for Sandwich tern in migration seasons (July-September and March-May) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters on migration	Proportion of immatures in UK western waters on migration	UK western Number adults	UK western Number immatures	UK western Total birds
Norway & Sweden	1990s	700	1400	882	0.05	0.05	70	44	114
Denmark	1990s	4500	9000	5670	0.03	0.03	270	170	440
Germany	1990s	9700	19400	12222	0.02	0.02	388	244	632
Netherlands	1990s	14500	29000	18270	0.01	0.01	290	183	473
Belgium	2000	1550	3100	1953	0.01	0.01	31	20	51
Ireland	2000	1800	3600	2268	0.3	0.3	1080	680	1760
Loch of Strathbeg	2013	0	0	0	0	0	0	0	0
Ythan Estuary	2013	565	1130	712	0	0	0	0	0
Forth Islands	2013	0	0	0	0	0	0	0	0
Farne Islands	2013	824	1648	1038	0	0	0	0	0
Coquet Island	2013	670	1340	844	0	0	0	0	0
North Norfolk Coast	2012	4135	8270	5210	0	0	0	0	0
Alde-Ore Estuary	2009	2	4	3	0	0	0	0	0
Foulness	2006	0	0	0	0	0	0	0	0
Chichester & Langstone Harb	2013	6	12	8	0	0	0	0	0
Solent & Southampton Water	2008	0	0	0	0	0	0	0	0
UK North Sea non-SPA colonies	2000	3500	7000	4410	0	0	0	0	0
Carlingford Lough	2013	0	0	0	1	0.7	0	0	0
Larne Lough	2013	257	514	324	1	0.7	514	227	741
Strangford Lough	2012	771	1542	971	1	0.7	1542	680	2222
Morecambe Bay	2011	1	2	1	1	0.7	2	1	3
Duddon Estuary	2012	1	2	1	1	0.7	2	1	3
Ynys Feurig, Cemlyn Bay	2009	0	0	0	1	0.7	0	0	0
UK Western non-SPA colonies	2000	1500	3000	1890	1	0.7	3000	1323	4323

Total overseas	2,129	1,341	3,470
Total UK	5,060	2,231	7,291
Total	7,189	3,572	10,761

Table 53. BDMPS for roseate tern in migration seasons (August-September and late-April-May) in 'UK East Coast & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK east coast & Channel waters on migration	Proportion of immatures in UK east coast & Channel waters on migration	UK east coast & Channel Number adults	UK east coast & Channel Number immatures	UK east coast & Channel Total birds
Germany, Netherlands, Belgium	2010	3	6	4	0.05	0.1	0	0	1
Ireland	2010	750	1500	1125	0.002	0.003	3	3	6
Forth Islands	2005-09	3	6	4	1	0.6	6	3	9
Farne Islands	2011	0	0	0	1	0.6	0	0	0
Coquet Island	2011	78	156	117	1	0.6	156	70	226
North Norfolk Coast	2010	0	0	0	1	0.6	0	0	0
Solent & Southampton Water	2009	0	0	0	1	0.6	0	0	0
UK North Sea non-SPA colonies	2010	3	6	4	1	0.6	6	3	9
Larne Lough	2011	0	0	0	0	0	0	0	0
Ynys Feurig, Cemlyn Bay & Skerries	2011	0	0	0	0	0	0	0	0
UK western non-SPA colonies	2010	0	0	0	0	0	0	0	0
Total overseas							3	4	7
Total UK							168	76	244
Total							171	80	251

Table 54. BDMPS for roseate tern in migration seasons (August-September and late-April-May) in 'north and west Scottish waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in N & W Scottish waters on migration	Proportion of immatures in N & W Scottish waters on migration	N & W Scottish Number adults	N & W Scottish Number immatures	N & W Scottish Total birds
Germany, Netherlands, Belgium	2010	3	6	4	0	0.001	0	0	0
Ireland	2010	750	1500	1125	0.0005	0.003	1	3	4
Forth Islands	2005-09	3	6	4	0	0	0	0	0
Farne Islands	2011	0	0	0	0	0	0	0	0
Coquet Island	2011	78	156	117	0	0	0	0	0
North Norfolk Coast	2010	0	0	0	0	0	0	0	0
Solent & Southampton Water	2009	0	0	0	0	0	0	0	0
UK North Sea non-SPA colonies	2010	3	6	4	0	0	0	0	0
Larne Lough	2011	0	0	0	0	0	0	0	0
Ynys Feurig, Cemlyn Bay & Skerries	2011	0	0	0	0	0	0	0	0
UK western non-SPA colonies	2010	0	0	0	0	0	0	0	0
Total overseas							1	3	4
Total UK							0	0	0
Total							1	3	4

Table 55. BDMPS for roseate tern in migration seasons (August-September and late-April-May) in 'Wales and west England waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in Wales & West England waters on migration	Proportion of immatures in Wales & West England waters on migration	Wales & West England Number adults	Wales & West England Number immatures	Wales & West England Total birds
Germany, Netherlands, Belgium	2010	3	6	4	0	0.001	0	0	0
Ireland	2010	750	1500	1125	0.95	0.6	1425	675	2100
Forth Islands	2005-09	3	6	4	0	0	0	0	0
Farne Islands	2011	0	0	0	0	0	0	0	0
Coquet Island	2011	78	156	117	0	0	0	0	0
North Norfolk Coast	2010	0	0	0	0	0	0	0	0
Solent & Southampton Water	2009	0	0	0	0	0	0	0	0
UK North Sea non-SPA colonies	2010	3	6	4	0	0	0	0	0
Larne Lough	2011	0	0	0	1	0.6	0	0	0
Ynys Feurig, Cemlyn Bay & Skerries	2011	0	0	0	1	0.6	0	0	0
UK western non-SPA colonies	2010	0	0	0	1	0.6	0	0	0
Total overseas							1,425	675	2,100
Total UK							0	0	0
Total							1,425	675	2,100

Table 56. BDMPS for common tern in migration seasons (late July-early September and April-May) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters on migration	Proportion of immatures in UK North Sea & Channel waters on migration	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Norway	1990s	15000	30000	20100	0.3	0.3	9000	6030	15030
Finland	1990s	50000	100000	67000	0.3	0.3	30000	20100	50100
Sweden	1990s	22000	44000	29480	0.3	0.3	13200	8844	22044
Baltic States	1990s	12750	25500	17085	0.3	0.3	7650	5126	12776
Germany & Denmark	1990s	10000	20000	13400	0.25	0.25	5000	3350	8350
Netherlands	1990s	19000	38000	25460	0.25	0.25	9500	6365	15865
Ireland	2000	2700	5400	3618	0.2	0.2	1080	724	1804
Cromarty Firth	2010	68	136	91	0.7	0.5	95	46	141
Inner Moray Firth	2013	0	0	0	0.7	0.5	0	0	0
Ythan Estuary, Sands of Forvie	2010	4	8	5	0.7	0.5	6	3	8
Forth Islands	2011	26	52	35	0.7	0.5	36	17	54
Imperial Dock Lock	2010	818	1636	1096	0.7	0.5	1145	548	1693
Farne Islands	2013	94	188	126	0.7	0.5	132	63	195
Coquet Island	2013	1041	2082	1395	0.7	0.5	1457	697	2155
The Wash	2013	221	442	296	0.7	0.5	309	148	457
North Norfolk Coast	2012	198	396	265	0.7	0.5	277	133	410
Breydon Water	2013	92	184	123	0.7	0.5	129	62	190
Foulness	2008	25	50	34	0.7	0.5	35	17	52
Dungeness to Pett Level	2013	79	158	106	0.7	0.5	111	53	164
Poole Harbour	2013	163	326	218	0.7	0.5	228	109	337
Solent & Southampton Water	2007	280	560	375	0.7	0.5	392	188	580
UK North Sea non-SPA colonies	2000	5500	11000	7370	0.7	0.5	7700	3685	11385
Glas Eileanan	2012	22	44	29	0.1	0.1	4	3	7
Carlingford Lough	2013	119	238	159	0.1	0.1	24	16	30

Larne Lough	2013	231	462	310	0.1	0.1	46	31	77
Lough Neagh & Lough Beg	2013	78	156	105	0.1	0.1	16	10	26
Strangford Lough	2013	352	704	472	0.1	0.1	70	47	118
The Dee Estuary	2013	165	330	221	0.1	0.1	33	22	55
Ribble & Alt Estuaries	2008	111	222	149	0.1	0.1	22	15	37
Ynys Feurig, Cemlyn Bay & Skerries	2011	178	356	239	0.1	0.1	36	24	59
UK western non-SPA colonies	2000	2100	4200	2814	0.1	0.1	420	281	701
Total overseas							75,430	50,539	125,969
Total UK							12,724	6,218	18,942
Total							88,154	56,757	144,911

Table 57. BDMPS for common tern in migration seasons (late July-early September and April-May) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters on migration	Proportion of immatures in UK western waters on migration	UK western waters Number adults	UK western waters Number immatures	UK western waters Total birds
Norway	1990s	15000	30000	20100	0.2	0.2	6000	4020	10020
Finland	1990s	50000	100000	67000	0.1	0.1	10000	6700	16700
Sweden	1990s	22000	44000	29480	0.1	0.1	4400	2948	7348
Baltic States	1990s	12750	25500	17085	0.1	0.1	2550	1708	4258
Germany & Denmark	1990s	10000	20000	13400	0.1	0.1	2000	1340	3340
Netherlands	1990s	19000	38000	25460	0.05	0.05	1900	1273	3173
Ireland	2000	2700	5400	3618	0.4	0.4	2160	1447	3607
Cromarty Firth	2010	68	136	91	0.3	0.2	41	18	59
Inner Moray Firth	2013	0	0	0	0.3	0.2	0	0	0
Ythan Estuary, Sands of Forvie	2010	4	8	5	0.3	0.2	2	1	3
Forth Islands	2011	26	52	35	0.3	0.2	16	7	23
Imperial Dock Lock	2010	818	1636	1096	0.3	0.2	491	219	710
Farne Islands	2013	94	188	126	0.3	0.2	56	25	82
Coquet Island	2013	1041	2082	1395	0.3	0.2	625	279	904
The Wash	2013	221	442	296	0.3	0.2	133	59	192
North Norfolk Coast	2012	198	396	265	0.3	0.2	119	53	172
Breydon Water	2013	92	184	123	0.3	0.2	55	25	80
Foulness	2008	25	50	34	0.3	0.2	15	7	22
Dungeness to Pett Level	2013	79	158	106	0.3	0.2	47	21	69
Poole Harbour	2013	163	326	218	0.3	0.2	98	44	141
Solent & Southampton Water	2007	280	560	375	0.3	0.2	168	75	243
UK North Sea non-SPA colonies	2000	5500	11000	7370	0.3	0.2	3300	1474	4774
Glas Eileanan	2012	22	44	29	0.9	0.6	40	18	57
Carlingford Lough	2013	119	238	159	0.9	0.6	214	96	310
Larne Lough	2013	231	462	310	0.9	0.6	416	186	602

Lough Neagh & Lough Beg	2013	78	156	105	0.9	0.6	140	63	203
Strangford Lough	2013	352	704	472	0.9	0.6	634	283	917
The Dee Estuary	2013	165	330	221	0.9	0.6	297	133	430
Ribble & Alt Estuaries	2008	111	222	149	0.9	0.6	200	89	289
Ynys Feurig, Cemlyn Bay & Skerries	2011	178	356	239	0.9	0.6	320	143	464
UK western non-SPA colonies	2000	2100	4200	2814	0.9	0.6	3780	1688	5468
Total overseas							29,010	19,437	48,447
Total UK							11,206	5,005	16,212
Total							40,216	24,442	64,659

Table 58. BDMPS for Arctic tern in migration seasons (July-early September and late April-May) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters on migration	Proportion of immatures in UK North Sea & Channel waters on migration	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Fennoscandia	1990s	131000	262000	151960	0.2	0.15	52400	22794	75194
Faroe	2012	7600	15200	8816	0.2	0.15	3040	1322	4362
Baltic States	1990s	8000	16000	9280	0.1	0.1	1600	928	2528
Ireland	2000	2500	5000	2900	0	0	0	0	0
Fetlar	2012	21	42	24	0.9	0.6	38	15	52
Foula	2013	20	40	23	0.9	0.6	36	14	50
Papa Stour	2000	1172	2344	1360	0.9	0.6	2110	816	2925
Mousa	2013	18	36	21	0.9	0.6	32	13	45
Sumburgh Head	2000	203	406	235	0.9	0.6	365	141	507
Fair Isle	2013	29	58	34	0.9	0.6	52	20	72
West Westray	2009	500	1000	580	0.9	0.6	900	348	1248
Papa Westray	2011	176	352	204	0.9	0.6	317	122	439
Rousay	2006	60	120	70	0.9	0.6	108	42	150
Auskerry	2013	750	1500	870	0.9	0.6	1350	522	1872
Pentland Firth Islands	2007	0	0	0	0.9	0.6	0	0	0
Forth Islands	2012	265	530	307	1	0.7	530	215	745
Farne Islands	2013	1921	3842	2228	1	0.7	3842	1560	5402
Coquet Island	2013	1224	2448	1420	1	0.7	2448	994	3442
UK North Sea non-SPA colonies	2000	26000	52000	30160	0.9	0.6	46800	18096	64896
Outer Ards	2013	60	120	70	0	0	0	0	0
Strangford Lough	2013	164	328	190	0	0	0	0	0
Ynys Feurig, Cemlyn Bay, Skerries	2011	550	1100	638	0	0	0	0	0
UK western non-SPA colonies	2000	15000	30000	17400	0	0	0	0	0

Total overseas	57,040	25,044	82,084
Total UK	58,928	22,917	81,846
Total	115,968	47,961	163,930

Table 59. BDMPS for Arctic tern in migration seasons (July-early September and late April-May) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters on migration	Proportion of immatures in UK western waters on migration	UK western waters Number adults	UK western waters Number immatures	UK western waters Total birds
Fennoscandia	1990s	131000	262000	151960	0.03	0.03	7860	4559	12419
Faroe	2012	7600	15200	8816	0.1	0.1	1520	882	2402
Baltic States	1990s	8000	16000	9280	0.02	0.02	320	186	506
Ireland	2000	2500	5000	2900	0.3	0.3	1500	870	2370
Fetlar	2012	21	42	24	0.1	0.1	4	2	7
Foula	2013	20	40	23	0.1	0.1	4	2	6
Papa Stour	2000	1172	2344	1360	0.1	0.1	234	136	370
Mousa	2013	18	36	21	0.1	0.1	4	2	6
Sumburgh Head	2000	203	406	235	0.1	0.1	41	24	64
Fair Isle	2013	29	58	34	0.1	0.1	6	3	9
West Westray	2009	500	1000	580	0.1	0.1	100	58	158
Papa Westray	2011	176	352	204	0.1	0.1	35	20	56
Rousay	2006	60	120	70	0.1	0.1	12	7	19
Auskerry	2013	750	1500	870	0.1	0.1	150	87	237
Pentland Firth Islands	2007	0	0	0	0.1	0.1	0	0	0
Forth Islands	2012	265	530	307	0	0.05	0	15	15
Farne Islands	2013	1921	3842	2228	0	0.05	0	111	111
Coquet Island	2013	1224	2448	1420	0	0.05	0	71	71
UK North Sea non-SPA colonies	2000	26000	52000	30160	0.1	0.1	5200	3016	8216
Outer Ards	2013	60	120	70	1	0.7	120	49	169
Strangford Lough	2013	164	328	190	1	0.7	328	133	461
Ynys Feurig, Cemlyn Bay, Skerries	2011	550	1100	638	1	0.7	1100	447	1547
UK western non-SPA colonies	2000	15000	30000	17400	1	0.7	30000	12180	42180
Total overseas							11,200	6,496	17,696

Total UK	37,338	16,364	53,702
Total	48,538	22,860	71,398

Table 60. BDMPS for little tern in migration seasons (late July to early September, and mid-April to May) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters on migration	Proportion of immatures in UK North Sea & Channel waters on migration	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Ireland	2000	200	400	224	0	0	0	0	0
Ythan Estuary, Sands of Forvie	2013	40	80	45	1	0.6	80	27	107
Firth of Tay & Eden Estuary	2007	1	2	1	1	0.6	2	1	3
Lindisfarne	2011	8	16	9	1	0.6	16	5	21
Northumbria Coast	2000	38	76	43	1	0.6	76	26	102
Teesmouth & Cleveland Est	2011	84	168	94	1	0.6	168	56	224
Gibraltar point	2011	12	24	13	1	0.6	24	8	32
Humber Flats, Marshes & Coast	2011	29	58	32	1	0.6	58	19	77
The Wash	2009	0	0	0	1	0.6	0	0	0
North Norfolk Coast	2011	409	818	458	1	0.6	818	275	1093
Alde-Ore Estuary	2009	0	0	0	1	0.6	0	0	0
Minsmere-Walberswick	2010	30	60	34	1	0.6	60	20	80
Great Yarmouth North Denes	2011	5	10	6	1	0.6	10	3	13
Foulness	2005	0	0	0	1	0.6	0	0	0
Dungeness to Pett Level	2013	11	22	12	1	0.6	22	7	29
Medway Estuary & Marshes	2009	18	36	20	1	0.6	36	12	48
Benacre to Easton Bavents	2011	45	90	50	1	0.6	90	30	120
Blackwater Estuary	2000	99	198	111	1	0.6	198	67	265
Colne Estuary	2011	0	0	0	1	0.6	0	0	0
Hamford Water	2011	45	90	50	1	0.6	90	30	120
Chesil Beach	2011	19	38	21	1	0.6	38	13	51
Chichester Harbour	2011	60	120	67	1	0.6	120	40	160
Pagham Harbour	2011	6	12	7	1	0.6	12	4	16
Solent & Southampton Water	2007	0	0	0	1	0.6	0	0	0

UK N Sea & Channel non-SPA colonies	2000	360	720	403	1	0.6	720	242	962
Monach Isles	2001	2	4	2	0	0	0	0	0
South Uist Machair & Lochs	2002	17	34	19	0	0	0	0	0
The Dee Estuary	2011	126	252	141	0	0	0	0	0
Morecambe Bay	2011	62	124	69	0	0	0	0	0
UK western non-SPA colonies	2000	200	400	224	0	0	0	0	0
Total overseas							0	0	0
Total UK							2,638	886	3,524
Total							2,638	886	3,524

Table 61. BDMPS for little tern in migration seasons (late July to early September, and mid-April to May) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters on migration	Proportion of immatures in UK western waters on migration	UK western waters Number adults	UK western waters Number immatures	UK western waters Total birds
Ireland	2000	200	400	224	0.95	0.6	380	134	514
Ythan Estuary, Sands of Forvie	2013	40	80	45	0	0	0	0	0
Firth of Tay & Eden Estuary	2007	1	2	1	0	0	0	0	0
Lindisfarne	2011	8	16	9	0	0	0	0	0
Northumbria Coast	2000	38	76	43	0	0	0	0	0
Teesmouth & Cleveland Est	2011	84	168	94	0	0	0	0	0
Gibraltar point	2011	12	24	13	0	0	0	0	0
Humber Flats, Marshes & Coast	2011	29	58	32	0	0	0	0	0
The Wash	2009	0	0	0	0	0	0	0	0
North Norfolk Coast	2011	409	818	458	0	0	0	0	0
Alde-Ore Estuary	2009	0	0	0	0	0	0	0	0
Minsmere-Walberswick	2010	30	60	34	0	0	0	0	0
Great Yarmouth North Denes	2011	5	10	6	0	0	0	0	0
Foulness	2005	0	0	0	0	0	0	0	0
Dungeness to Pett Level	2013	11	22	12	0	0	0	0	0
Medway Estuary & Marshes	2009	18	36	20	0	0	0	0	0
Benacre to Easton Bavents	2011	45	90	50	0	0	0	0	0
Blackwater Estuary	2000	99	198	111	0	0	0	0	0
Colne Estuary	2011	0	0	0	0	0	0	0	0
Hamford Water	2011	45	90	50	0	0	0	0	0
Chesil Beach	2011	19	38	21	0	0	0	0	0
Chichester Harbour	2011	60	120	67	0	0	0	0	0
Pagham Harbour	2011	6	12	7	0	0	0	0	0
Solent & Southampton Water	2007	0	0	0	0	0	0	0	0
UK N Sea & Channel non-SPA colonies	2000	360	720	403	0	0	0	0	0

Monach Isles	2001	2	4	2	1	0.6	4	1	5
South Uist Machair & Lochs	2002	17	34	19	1	0.6	34	11	45
The Dee Estuary	2011	126	252	141	1	0.6	252	85	337
Morecambe Bay	2011	62	124	69	1	0.6	124	42	166
UK western non-SPA colonies	2000	200	400	224	1	0.6	400	134	534
Total overseas							380	134	514
Total UK							814	274	1,088
Total							1,194	408	1,602

Table 62. BDMPS for common guillemot in non-breeding season (August to February) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in non-breeding season	Proportion of immatures in UK North Sea & Channel waters in non-breeding season	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Faroe Islands	c2000	100,000	200,000	148000	0.1	0.2	20000	29600	49600
Norway	c2000	100,000	200,000	148000	0.05	0.2	10000	29600	39600
Hermaness, Saxavord & Valla Field SPA	2009	4620	9,240	6838	0.7	0.6	6468	4103	10571
Foula SPA	2007	16615	33,230	24590	0.7	0.6	23261	14754	38015
Noss SPA	2009	14783	29,566	21879	0.7	0.6	20696	13127	33824
Sumburgh SPA	2010	4762	9,524	7048	0.7	0.6	6667	4229	10896
Fair Isle SPA	2010	13066	26,132	19338	0.7	0.6	18292	11603	29895
West Westray SPA	2007	33900	67,800	50172	0.7	0.6	47460	30103	77563
Calf of Eday SPA	2006	6300	12,600	9324	0.7	0.6	8820	5594	14414
Rousay SPA	2009	6200	12,400	9176	0.7	0.6	8680	5506	14186
Marwick Head SPA	2012	11097	22,194	16424	0.7	0.6	15536	9854	25390
Hoy SPA	2007	6300	12,600	9324	0.7	0.6	8820	5594	14414
Copinsay SPA	2012	5607	11,214	8298	0.7	0.6	7850	4979	12829
North Caithness Cliffs SPA	2000	47000	94,000	69560	0.7	0.6	65800	41736	107536
East Caithness Cliffs SPA	1999	106500	213,000	157620	0.7	0.6	149100	94572	243672
Troup, Pennan & Lion's Heads SPS	2007	10938	21,876	16188	0.7	0.6	15313	9713	25026
Buchan Ness to Collieston Coast SPA	2007	12928	25,856	19133	0.8	0.7	20685	13393	34078
Fowlsheugh SPA	2012	30100	60,200	44548	0.8	0.7	48160	31184	79344
Forth Islands SPA	2011	14674	29,348	21718	0.9	0.8	26413	17374	43787
St Abb's Head to Fast Castle SPA	2013	22103	44,206	32712	0.9	0.8	39785	26170	65955
Farne Islands SPA	2013	33532	67,064	49627	0.9	0.8	60358	39702	100059

Flamborough and Filey Coast pSPA	2008	39641	79,282	58669	0.9	0.8	71354	46935	118289
Germany and Denmark	2005	5,000	10,000	7400	0.2	0.4	2000	2960	4960
UK North Sea non-SPA populations	2000	147000	294,000	217560	0.8	0.6	235200	130536	365736
Sule Skerry & Sule Stack SPA	1998	7633	15,266	11297	0.05	0.1	763	1130	1893
North Rona & Sula Sgeir SPA	2012	5000	10,000	7400	0.05	0.1	500	740	1240
Cape Wrath SPA	2000	27359	54,718	40491	0.05	0.1	2736	4049	6785
Handa SPA	2011	37993	75,986	56230	0.05	0.1	3799	5623	9422
Shiant Isles SPA	2008	5148	10,296	7619	0.05	0.1	515	762	1277
Flannan Isles SPA	1999	9807	19,614	14514	0.05	0.1	981	1451	2432
St Kilda SPA	1999	15700	31,400	23236	0.05	0.1	1570	2324	3893
Canna & Sanday SPA	1999	3913	7,826	5791	0.05	0.1	391	579	970
Rum SPA	2000	1644	3,288	2433	0.05	0.1	164	243	408
Mingulay & Berneray SPA	2009	13527	27,054	20020	0.05	0.1	1353	2002	3355
North Colonsay & western cliffs SPA	2000	13500	27,000	20000	0	0.05	0	1000	1000
Ailsa Craig SPA	2013	5247	10,494	7766	0	0.05	0	388	388
Rathlin Island SPA	2011	87398	174,796	129349	0	0.05	0	6467	6467
Skomer & Skokholm SPA	2013	16300	32,600	24124	0.05	0.1	1630	2412	4042
UK West coast non-SPA populations	2000	79000	158,000	116920	0.03	0.08	4740	9354	14094
Total overseas							32,000	62,160	94,160
Total UK							923,860	599,286	1,523,146
Total							955,860	661,446	1,617,306

Table 63. BDMPS for common guillemot in non-breeding season (August to February) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in non-breeding season	Proportion of immatures in UK western waters in non-breeding season	UK western waters number of adults	UK western waters number of immatures	UK western waters Total birds
Faroe Islands	c2000	100,000	200,000	148000	0.05	0.1	10000	14800	24800
Norway	c2000	100,000	200,000	148000	0.01	0.05	2000	7400	9400
Hermaness, Saxavord & Valla Field SPA	2009	4620	9,240	6838	0.02	0.05	185	342	527
Foula SPA	2007	16615	33,230	24590	0.02	0.05	665	1230	1894
Noss SPA	2009	14783	29,566	21879	0.02	0.05	591	1094	1685
Sumburgh SPA	2010	4762	9,524	7048	0.02	0.05	190	352	543
Fair Isle SPA	2010	13066	26,132	19338	0.02	0.05	523	967	1490
West Westray SPA	2007	33900	67,800	50172	0.02	0.05	1356	2509	3865
Calf of Eday SPA	2006	6300	12,600	9324	0.02	0.05	252	466	718
Rousay SPA	2009	6200	12,400	9176	0.02	0.05	248	459	707
Marwick Head SPA	2012	11097	22,194	16424	0.02	0.05	444	821	1265
Hoy SPA	2007	6300	12,600	9324	0.02	0.05	252	466	718
Copinsay SPA	2012	5607	11,214	8298	0.02	0.05	224	415	639
North Caithness Cliffs SPA	2000	47000	94,000	69560	0.02	0.05	1880	3478	5358
East Caithness Cliffs SPA	1999	106500	213,000	157620	0	0	0	0	0
Troup, Pennan & Lion's Heads SPS	2007	10938	21,876	16188	0	0	0	0	0
Buchan Ness to Collieston Coast SPA	2007	12928	25,856	19133	0	0	0	0	0
Fowlsheugh SPA	2012	30100	60,200	44548	0	0	0	0	0
Forth Islands SPA	2011	14674	29,348	21718	0	0	0	0	0
St Abb's Head to Fast Castle SPA	2013	22103	44,206	32712	0	0	0	0	0
Farne Islands SPA	2013	33532	67,064	49627	0	0	0	0	0
Flamborough and Filey Coast pSPA	2008	39641	79,282	58669	0	0	0	0	0
Germany and Denmark	2005	5,000	10,000	7400	0	0	0	0	0
North Sea UK non-SPA populations	2000	147000	294,000	217560	0.01	0.02	2940	4351	7291

Sule Skerry & Sule Stack SPA	1998	7633	15,266	11297	0.95	0.9	14503	10167	24670
North Rona & Sula Sgeir SPA	2012	5000	10,000	7400	0.95	0.9	9500	6660	16160
Cape Wrath SPA	2000	27359	54,718	40491	0.95	0.9	51982	36442	88424
Handa SPA	2011	37993	75,986	56230	0.95	0.9	72187	50607	122793
Shiant Isles SPA	2008	5148	10,296	7619	0.95	0.9	9781	6857	16638
Flannan Isles SPA	1999	9807	19,614	14514	0.95	0.9	18633	13063	31696
St Kilda SPA	1999	15700	31,400	23236	0.95	0.9	29830	20912	50742
Canna & Sanday SPA	1999	3913	7,826	5791	0.95	0.9	7435	5212	12647
Rum SPA	2000	1644	3,288	2433	0.95	0.9	3124	2190	5313
Mingulay & Berneray SPA	2009	13527	27,054	20020	0.95	0.9	25701	18018	43719
North Colonsay and western cliffs SPA	2000	13500	27,000	20000	1	0.95	27000	19000	46000
Ailsa Craig SPA	2013	5247	10,494	7766	1	0.95	10494	7377	17871
Rathlin Island SPA	2011	87398	174,796	129349	1	0.95	174796	122882	297678
Skomer & Skokholm SPA	2013	16300	32,600	24124	0.9	0.8	29340	19299	48639
West coast UK non-SPA populations	2000	79000	158,000	116920	0.95	0.9	150100	105228	255328
Total overseas							12,000	22,200	34,200
Total UK							644,156	460,864	1,105,020
Total							656,156	483,064	1,139,220

Table 64. BDMPS for razorbill in migration seasons (August to October, and January to March) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in migration	Proportion of immatures in UK North Sea & Channel waters in migration	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Russia	1990s	3500	7000	5250	0.05	0.1	350	525	875
Iceland	2008	315400	630800	473100	0.3	0.4	189240	189240	378480
Norway	1990s	30300	60600	45450	0.2	0.5	12120	22725	34845
Denmark, Finland, Sweden	1990s	16000	32000	24000	0.1	0.3	3200	7200	10400
Faroe	2012	4500	9000	6750	0.5	0.5	4500	3375	7875
Foula	2007	375	750	562	0.95	0.9	712	506	1219
Fair Isle	2010	915	1830	1372	0.95	0.9	1738	1235	2974
West Westray	2007	550	1100	825	0.95	0.9	1045	742	1788
North Caithness Cliffs	2000	1700	3400	2550	0.95	0.9	3230	2295	5525
East Caithness Cliffs	1999	12500	25000	18750	1	0.9	25000	16875	41875
Troup, Pennan & Lions	2007	1743	3486	2614	1	0.9	3486	2353	5839
Fowlsheugh	2012	3524	7048	5286	1	0.9	7048	4757	11805
Forth Islands	2012	2625	5250	3938	1	0.9	5250	3544	8794
St Abbs to Fast Castle	2013	1219	2438	1828	1	0.9	2438	1646	4084
Flamborough & Filey	2008	10001	20002	15002	1	0.9	20002	13501	33503
UK North Sea non-SPA colonies	2000	10000	20000	15000	1	0.9	20000	13500	33500
North Rona & Sula Sgeir	1998	1089	2178	1634	0.02	0.05	44	82	125
Cape Wrath	2000	2090	4180	3135	0.02	0.05	84	157	240
Handa	2010	5165	10330	7748	0.02	0.05	207	387	594
St Kilda	1999	1700	3400	2550	0.02	0.05	68	128	196
Shiants	2008	4248	8496	6372	0.02	0.05	170	319	489
Flannan Islands	1998	1051	2102	1576	0.02	0.05	42	79	121
Mingulay & Berneray	2009	10111	20222	15166	0.02	0.05	404	758	1163
Rathlin Island	2011	15393	30786	23090	0.02	0.05	616	1154	1770

Skomer & Skokholm	2013	6001	12002	9002	0.02	0.05	240	450	690
UK Western non-SPA colonies	2000	10000	20000	15000	0.02	0.05	400	750	1150
Ireland	2000	17000	34000	25500	0.02	0.05	680	1275	1955
France	2000	25	50	38	0.01	0.02	0	1	1
Total overseas							210,090	224,341	434,431
Total UK							92,224	65,219	157,443
Total							302,314	289,560	591,874

Table 65. BDMPS for razorbill in migration seasons (August to October, and January to March) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in migration	Proportion of immatures in UK western waters in migration	UK western waters number of adults	UK western waters number of immatures	UK western waters Total birds
Russia	1990s	3500	7000	5250	0.05	0.1	350	525	875
Iceland	2008	315400	630800	473100	0.3	0.4	189240	189240	378480
Norway	1990s	30300	60600	45450	0.1	0.3	6060	13635	19695
Denmark, Finland, Sweden	1990s	16000	32000	24000	0.05	0.1	1600	2400	4000
Faroe	2012	4500	9000	6750	0.5	0.5	4500	3375	7875
Foula	2007	375	750	562	0.05	0.05	38	28	66
Fair Isle	2010	915	1830	1372	0.05	0.05	92	69	160
West Westray	2007	550	1100	825	0.05	0.05	55	41	96
North Caithness Cliffs	2000	1700	3400	2550	0.05	0.05	170	128	298
East Caithness Cliffs	1999	12500	25000	18750	0	0.02	0	375	375
Troup, Pennan & Lions	2007	1743	3486	2614	0	0.02	0	52	52
Fowlsheugh	2012	3524	7048	5286	0	0.02	0	106	106
Forth Islands	2012	2625	5250	3938	0	0.02	0	79	79
St Abbs to Fast Castle	2013	1219	2438	1828	0	0.02	0	37	37
Flamborough & Filey	2008	10001	20002	15002	0	0.02	0	300	300
UK North Sea non-SPA colonies	2000	10000	20000	15000	0	0.02	0	300	300
North Rona & Sula Sgeir	1998	1089	2178	1634	0.98	0.9	2134	1470	3605
Cape Wrath	2000	2090	4180	3135	0.98	0.9	4096	2822	6918
Handa	2010	5165	10330	7748	0.98	0.9	10123	6973	17096
St Kilda	1999	1700	3400	2550	0.98	0.9	3332	2295	5627
Shiants	2008	4248	8496	6372	0.98	0.9	8326	5735	14061
Flannan Islands	1998	1051	2102	1576	0.98	0.9	2060	1419	3479
Mingulay & Berneray	2009	10111	20222	15166	0.98	0.9	19818	13650	33467
Rathlin Island	2011	15393	30786	23090	0.98	0.9	30170	20781	50951
Skomer & Skokholm	2013	6001	12002	9002	0.98	0.9	11762	8101	19863

UK Western non-SPA colonies	2000	10000	20000	15000	0.98	0.9	19600	13500	33100
Ireland	2000	17000	34000	25500	0.1	0.1	3400	2550	5950
France	2000	25	50	38	0.05	0.05	2	2	4
Total overseas							205,152	211,727	416,879
Total UK							111,776	78,259	190,035
Total							316,928	289,986	606,914

Table 66. BDMPS for razorbill in winter (November and December) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in winter	Proportion of immatures in UK North Sea & Channel waters in winter	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Russia	1990s	3500	7000	5250	0.01	0.02	70	105	175
Iceland	2008	315400	630800	473100	0.1	0.2	63080	94620	157700
Norway	1990s	30300	60600	45450	0.05	0.1	3030	4545	7575
Denmark, Finland, Sweden	1990s	16000	32000	24000	0.02	0.05	640	1200	1840
Faroe	2012	4500	9000	6750	0.3	0.3	2700	2025	4725
Foula	2007	375	750	562	0.3	0.1	225	56	281
Fair Isle	2010	915	1830	1372	0.3	0.1	549	137	686
West Westray	2007	550	1100	825	0.3	0.1	330	82	412
North Caithness Cliffs	2000	1700	3400	2550	0.3	0.1	1020	255	1275
East Caithness Cliffs	1999	12500	25000	18750	0.3	0.1	7500	1875	9375
Troup, Pennan & Lions	2007	1743	3486	2614	0.3	0.1	1046	261	1307
Fowlsheugh	2012	3524	7048	5286	0.3	0.1	2114	529	2643
Forth Islands	2012	2625	5250	3938	0.3	0.1	1575	394	1969
St Abbs to Fast Castle	2013	1219	2438	1828	0.3	0.1	731	183	914
Flamborough & Filey	2008	10001	20002	15002	0.3	0.1	6001	1500	7501
UK North Sea non-SPA colonies	2000	10000	20000	15000	0.3	0.1	6000	1500	7500
North Rona & Sula Sgeir	1998	1089	2178	1634	0.1	0.05	218	82	299
Cape Wrath	2000	2090	4180	3135	0.1	0.05	418	157	575
Handa	2010	5165	10330	7748	0.1	0.05	1033	387	1420
St Kilda	1999	1700	3400	2550	0.1	0.05	340	128	468
Shiants	2008	4248	8496	6372	0.1	0.05	850	319	1168
Flannan Islands	1998	1051	2102	1576	0.1	0.05	210	79	289
Mingulay & Berneray	2009	10111	20222	15166	0.1	0.05	2022	758	2781
Rathlin Island	2011	15393	30786	23090	0.05	0	1539	0	1539

Skomer & Skokholm	2013	6001	12002	9002	0.05	0	600	0	600
UK Western non-SPA colonies	2000	10000	20000	15000	0.1	0.05	2000	750	2750
Ireland	2000	17000	34000	25500	0.01	0.02	340	510	850
France	2000	25	50	38	0.05	0.05	2	2	4
Total overseas							69,862	103,007	172,869
Total UK							36,321	9,432	45,753
Total							106,183	112,439	218,622

Table 67. BDMPS for razorbill in winter (November and December) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in winter	Proportion of immatures in UK western waters in winter	UK western waters number of adults	UK western waters number of immatures	UK western waters Total birds
Russia	1990s	3500	7000	5250	0.01	0.02	70	105	175
Iceland	2008	315400	630800	473100	0.2	0.3	126160	141930	268090
Norway	1990s	30300	60600	45450	0.05	0.1	3030	4545	7575
Denmark, Finland, Sweden	1990s	16000	32000	24000	0.02	0.05	640	1200	1840
Faroe	2012	4500	9000	6750	0.3	0.3	2700	2025	4725
Foula	2007	375	750	562	0.01	0.02	8	11	19
Fair Isle	2010	915	1830	1372	0.01	0.02	18	27	46
West Westray	2007	550	1100	825	0.01	0.02	11	16	28
North Caithness Cliffs	2000	1700	3400	2550	0.01	0.02	34	51	85
East Caithness Cliffs	1999	12500	25000	18750	0.01	0.02	250	375	625
Troup, Pennan & Lions	2007	1743	3486	2614	0.01	0.02	35	52	87
Fowlsheugh	2012	3524	7048	5286	0.01	0.02	70	106	176
Forth Islands	2012	2625	5250	3938	0.01	0.02	52	79	131
St Abbs to Fast Castle	2013	1219	2438	1828	0.01	0.02	24	37	61
Flamborough & Filey	2008	10001	20002	15002	0.01	0.02	200	300	500
UK North Sea non-SPA colonies	2000	10000	20000	15000	0.01	0.02	200	300	500
North Rona & Sula Sgeir	1998	1089	2178	1634	0.4	0.1	871	163	1035
Cape Wrath	2000	2090	4180	3135	0.4	0.1	1672	314	1986
Handa	2010	5165	10330	7748	0.4	0.1	4132	775	4907
St Kilda	1999	1700	3400	2550	0.4	0.1	1360	255	1615
Shiants	2008	4248	8496	6372	0.4	0.1	3398	637	4036
Flannan Islands	1998	1051	2102	1576	0.4	0.1	841	158	998
Mingulay & Berneray	2009	10111	20222	15166	0.4	0.1	8089	1517	9605
Rathlin Island	2011	15393	30786	23090	0.4	0.1	12314	2309	14623
Skomer & Skokholm	2013	6001	12002	9002	0.3	0.1	3601	900	4501

UK Western non-SPA colonies	2000	10000	20000	15000	0.3	0.1	6000	1500	7500
Ireland	2000	17000	34000	25500	0.1	0.1	3400	2550	5950
France	2000	25	50	38	0.05	0.05	2	2	4
Total overseas							136,002	152,357	288,359
Total UK							43,181	9,882	53,063
Total							179,183	162,239	341,422

Table 68. BDMPS for Atlantic puffin in non-breeding season (mid-August to March) in 'UK North Sea & Channel waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK North Sea & Channel waters in non-breeding season	Proportion of immatures in UK North Sea & Channel waters in non-breeding season	UK N Sea & Channel Number adults	UK N Sea & Channel Number immatures	UK N Sea & Channel Total birds
Norway	2000	1750000	3500000	3640000	0.001	0.003	3500	10920	14420
Faroe	2012	550000	1100000	1144000	0.04	0.01	44000	11440	55440
Ireland	2000	20000	40000	41600	0	0	0	0	0
France	2000	257	514	535	0.05	0.02	26	11	36
Hermaness, Saxavord	2002	23661	47322	49215	0.15	0.02	7098	984	8083
Foula	2000	22500	45000	46800	0.15	0.02	6750	936	7686
Noss	2007	802	1604	1668	0.15	0.02	241	33	274
Fair Isle	2012	10706	21412	22268	0.15	0.02	3212	445	3657
Hoy	2000	3500	7000	7280	0.15	0.02	1050	146	1196
North Caithness Cliffs	2000	976	1952	2030	0.15	0.02	293	41	333
East Caithness Cliffs	1999	274	548	570	0.15	0.02	82	11	94
Forth Islands	2008-10	62231	124462	129440	0.5	0.02	62231	2589	64820
Farne Islands	2013	39962	79924	83121	0.5	0.02	39962	1662	41624
Coquet Island	2013	12344	24688	25676	0.5	0.02	12344	514	12858
Flamborough & Filey	2008	958	1916	1993	0.5	0.02	958	40	998
UK N Sea non-SPA colonies	2000	35000	70000	72800	0.25	0.02	17500	1456	18956
Cape Wrath	2000	1602	3204	3332	0.001	0.001	3	3	7
North Rona & Sula Sgeir	2001	5442	10884	11319	0.001	0.001	11	11	22
Sule Skerry & Sule Stack	1998	59471	118942	123700	0.001	0.001	119	124	243
St Kilda	2000	142264	284528	295909	0.001	0.001	285	296	580
Shiant Isles	2000	65170	130340	135554	0.001	0.001	130	136	266
Flannan Isles	2001	15600	31200	32448	0.001	0.001	31	32	63
Canna & Sanday	1999	945	1890	1966	0.001	0.001	2	2	4
Mingulay & Berneray	2009	3126	6252	6502	0.001	0.001	6	7	13

Rathlin Island	2011	695	1390	1446	0.001	0.001	1	1	3
Skomer &Skokholm	2013	24114	48228	50157	0.001	0.001	48	50	98
UK western non-SPA colonies	2000	45000	90000	93600	0.001	0.001	90	94	184
Overseas total							47,526	22,371	69,896
UK total							152,448	9,613	162,061
Total							199,974	31,984	231,957

Table 69. BDMPS for Atlantic puffin in non-breeding season (mid-August to March) in 'UK western waters'.

Population	Most recent count	Pairs	Breeding adults	Immatures	Proportion of adults in UK western waters in non-breeding season	Proportion of immatures in UK western waters in non-breeding season	UK western waters Number adults	UK western waters Number immatures	UK western waters Total birds
Norway	2000	1750000	3500000	3640000	0.002	0.001	7000	3640	10640
Faroe	2012	550000	1100000	1144000	0.07	0.02	77000	22880	99880
Ireland	2000	20000	40000	41600	0.1	0.1	4000	4160	8160
France	2000	257	514	535	0.01	0.01	5	5	10
Hermaness, Saxavord	2002	23661	47322	49215	0.08	0.02	3786	984	4770
Foula	2000	22500	45000	46800	0.08	0.02	3600	936	4536
Noss	2007	802	1604	1668	0.08	0.02	128	33	162
Fair Isle	2012	10706	21412	22268	0.08	0.02	1713	445	2158
Hoy	2000	3500	7000	7280	0.08	0.02	560	146	706
North Caithness Cliffs	2000	976	1952	2030	0.08	0.02	156	41	197
East Caithness Cliffs	1999	274	548	570	0.08	0.02	44	11	55
Forth Islands	2008-10	62231	124462	129440	0.07	0.02	8712	2589	11301
Farne Islands	2013	39962	79924	83121	0.07	0.02	5595	1662	7257
Coquet Island	2013	12344	24688	25676	0.07	0.02	1728	514	2242
Flamborough & Filey	2008	958	1916	1993	0.07	0.02	134	40	174
UK N Sea non-SPA colonies	2000	35000	70000	72800	0.07	0.02	4900	1456	6356
Cape Wrath	2000	1602	3204	3332	0.18	0.02	577	67	643
North Rona & Sula Sgeir	2001	5442	10884	11319	0.18	0.02	1959	226	2186
Sule Skerry & Sule Stack	1998	59471	118942	123700	0.18	0.02	21410	2474	23884
St Kilda	2000	142264	284528	295909	0.18	0.02	51215	5918	57133
Shiant Isles	2000	65170	130340	135554	0.18	0.02	23461	2711	26172
Flannan Isles	2001	15600	31200	32448	0.18	0.02	5616	649	6265
Canna & Sanday	1999	945	1890	1966	0.18	0.02	340	39	380
Mingulay & Berneray	2009	3126	6252	6502	0.18	0.02	1125	130	1255

Rathlin Island	2011	695	1390	1446	0.18	0.02	250	29	279
Skomer &Skokholm	2013	24114	48228	50157	0.18	0.02	8681	1003	9684
UK western non-SPA colonies	2000	45000	90000	93600	0.18	0.02	16200	1872	18072
Overseas total							88,005	30,685	118,690
UK total							161,891	23,976	185,867
Total							249,896	54,661	304,557

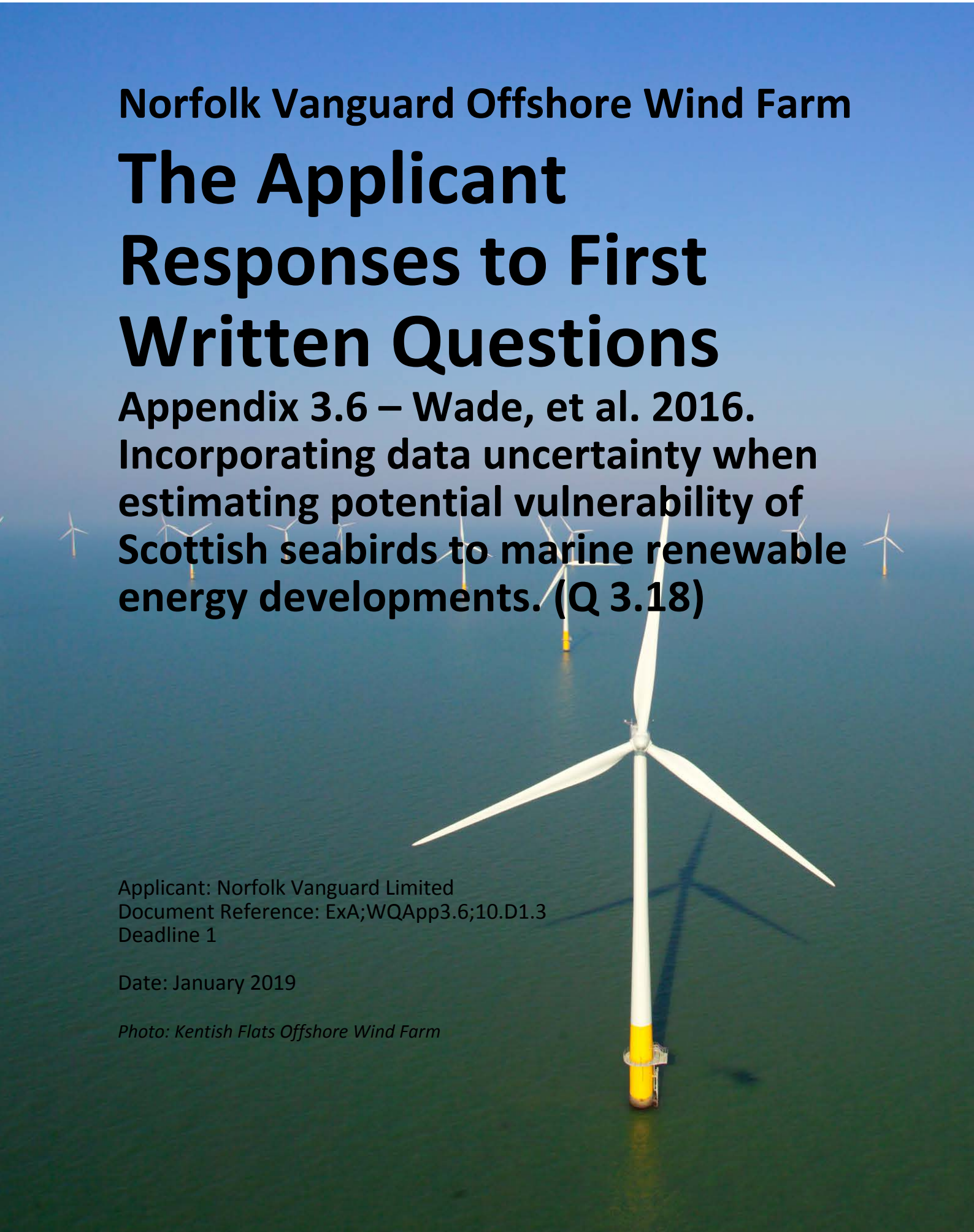
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The Applicant Responses to First Written Questions

Appendix 3.6 – Wade, et al. 2016. Incorporating data uncertainty when estimating potential vulnerability of Scottish seabirds to marine renewable energy developments. (Q 3.18)



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Short Communication

Incorporating data uncertainty when estimating potential vulnerability of Scottish seabirds to marine renewable energy developments

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ABSTRACT

The effects of marine renewable energy developments (MREDs) on seabirds are uncertain because of the relative infancy of the industry. This uncertainty can delay the consenting process as regulators adopt a precautionary approach. This study uses novel methods to demonstrate uncertainty in two indices that ranked the vulnerability of seabird populations to MREDs. The study also consolidates recently available data with information from the two indices to consider developments in our understanding of how seabirds respond to MREDs and to present up-to-date vulnerability predictions. Results indicate greater uncertainty in data regarding displacement caused by vessels and/or helicopters, and use of tidal races by seabirds, than in data regarding the percentage of flight overlapping with wind turbine blades and the level of displacement caused by structures. Results also indicate varying uncertainty among species. Overall vulnerability rankings remained broadly the same, with some minor changes. The uncertainty indices highlight areas lacking data, identify robust predictions, and indicate where particular caution in interpreting vulnerability indices should be adopted. They are a useful tool to inform impact assessment and identify strategic research and monitoring priorities.

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1. Introduction

Marine renewable energy developments (MREDs) are increasing worldwide to provide an alternative to fossil fuels, increase energy security and mitigate against climatic change [6,7,19]. Scotland has valuable marine renewable energy resources [1,21] and has developed a marine plan, including offshore wind, wave and tidal-stream technologies, to contribute to generating 100% of Scotland's electricity through renewable sources by 2020 [19]. Scotland is internationally important for seabirds [2,16], with special protection areas (SPAs) designated to safeguard breeding colonies [9,18,20]. With several leased and proposed Scottish MRED sites located close to SPAs for breeding seabirds, consideration of the potential consequences for seabirds is necessary.

The effects of MREDs on seabirds are uncertain because of the relative infancy of the industry, the early stage of some environmental monitoring programmes [25] and a limited ability to effectively monitor post-construction effects [13,14,17]. Uncertainty over effects can delay the consenting process as regulators adopt a precautionary approach [15]; for example, by using avoidance rates that may overestimate collision risk. In the absence of information regarding specific effects of MREDs on seabirds, a common approach is to use existing knowledge of seabird behaviour and ecology to derive estimates of seabird vulnerability (e.g. [3–5]). Uncertainty in the contributing data is, however, rarely presented, but is vital information, as the reliability of results and confidence in interpretations can be affected by the quality, quantity and relevance of contributing data [15]. These measures of data uncertainty identify where evidence supporting vulnerability rankings is more robust; where caution in interpreting results may be required; and where additional monitoring and research could prove beneficial [22].

Using Furness et al. [3,4] as examples, this study developed novel methods to incorporate uncertainty into indices ranking the vulnerability of Scottish seabird populations to MREDs. Furness et al. [3,4] developed four indices ranking vulnerability to i) collision with offshore wind turbines, ii) displacement caused by offshore wind farms, iii) wave energy, and iv) tidal-stream energy developments. These indices have been used by MRED regulators and developers during initial scoping and impact assessment (e.g.

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[8]) but measures of uncertainty in data contributing to rankings were not explicitly included. This study develops uncertainty indices to aid transparent and consistent application of vulnerability index predictions. Recently available data were consolidated with information in Furness et al. [3,4], to account for new developments in our understanding of how seabirds respond to i) structures and ii) vessels and helicopters, and to incorporate a reduced risk of collision with offshore wind turbines for species displaced by structures. The development of uncertainty indices and modified vulnerability indices more accurately represent the risks posed by MREDs to seabirds.

2. Methods

2.1. Calculating uncertainty

Four vulnerability factors were identified as important in driving seabird vulnerability to MREDs [3,4]: i) percentage of flight overlapping with wind turbine blades, ii) displacement caused by structures, iii) displacement caused by vessels and/or helicopters, and iv) use of tidal races. The quality, quantity and relevance of data contributing to these factors were assessed for each of 38 Scottish seabird species to estimate data uncertainty (see Supplementary Table 1 for scientific names). Data uncertainty was assessed using five criteria, with greater scores reflecting a greater quantity and quality of data, and therefore indicating lower levels of uncertainty:

1) **Species Score**: Did data refer to the target species or a related species? Species were scored 3 if $\geq$ 50% of data sources referred to the target species, 2 if data referred to a related species or to higher taxa, and 1 if no published data were available.
2) **Number of Sites**: How many sites contributed data?
3) **Number of Studies**: How many studies are included?
4) **Mean Years**: What was the mean period of years over which data were collected?
5) **Method Score**: What level of uncertainty was associated with the methods used to collect data? For a full explanation of the Methods Score, Method Categories and associated Uncertainty Levels see Section 2.1.1 and Table 1.

The five criteria scores derived for each species, in each vulnerability factor, are shown in Supplementary Tables 2–5.

2.1.1. Method Score

To generate a Method Score for each species, in each vulnerability factor, the number of studies in each Method Category (with different Method Categories considered relevant for the four vulnerability factors; Table 1) were multiplied by the Uncertainty Score under which the Method Category was located (Table 1). Greater weight was given to studies using more reliable and robust methods; for example, before-after-control-impact studies and studies collecting data on flight altitudes using bird-borne GPS devices. These more reliable methods were associated with greater scores to reflect a greater quality of data, and therefore a corresponding lower Uncertainty Level (Table 1; Eqs. (1) and (2)). The Method Score reflects the reliability of the methods used in all studies considered for each species in each vulnerability factor, and the uncertainty inherent in those data. Eq. (1) was used to

Table 1
Uncertainty Levels and Scores indicating the level of uncertainty associated with data contributing to species vulnerability rankings. Five categories with associated ranking scores indicate the level of uncertainty: very high (score 1), high (score 2), moderate (score 3), low (score 4) and very low uncertainty (score 5). Capital letters in brackets refer to the Method Categories included in Eqs. (1) and (2). The table indicates the Uncertainty Level and Score assigned to each Method Category and outlines the range of values included in the Combined Score at each Uncertainty Level, which differs among vulnerability factors. Greater scores reflect a greater quantity and quality of data, and therefore correspond to lower levels of uncertainty.

Vulnerability factor	Vulnerability factor attributes	Uncertainty Level (Uncertainty Scores)				
		Very high (1)	**High (2)**	**Moderate (3)**	**Low (4)**	**Very low (5)**
% Time flying at turbine height	**Method Category**	Anecdotal observation (or unknown method) **(A)**	Observations not recorded in the presence of turbines (indirect study 2) **(B)**	Observations recorded in the presence of turbines (indirect study 1) **(C)**	Study combining results from 5 or more studies/ sites to produce modelled flight information **(D)**	GPS or radar (direct study) **(E)**
	Combined Score	0.0 – 28.5	29.0 – 56.5	57.0 – 84.5	85.0 – 112.5	113.0–140.5
Disturbance by structures	**Method Category**	Anecdotal observation (or unknown method) **(A)**	Observation **(B)**	Before-After- Control-Impact study (BACI) **(C)**		
	Combined Score	0.0–12.5	13.0–24.5	25.0–36.5	37.0–48.5	49.0–60.5
Disturbance by vessel and/or helicopter activity	**Method Category**	Anecdotal observation (or unknown method) **(A)**	Observation **(B)**	BACI or experimental method **(C)**		
	Combined Score	0.0–8.5	9.0–16.5	17.0–24.5	25.0–32.5	33.0–40.5
Use of tidal races	**Method Category**	Anecdotal observation (or unknown method) **(A)**	Observation without current data **(B)**	Observation with modelled or inferred current data **(C)**	Study combining results from 5 or more studies/ sites with modelled or inferred current data **(D)**	Observation with concurrent current data **(E)**
	Combined Score	0.0–8.5	9.0–16.5	17.0–24.5	25.0–32.5	33.0–41.5

calculate uncertainty associated with each species in the vulnerability factors 'percentage of flight overlapping with wind turbine blades', and 'use of tidal races'. Eq. (2) was used to calculate uncertainty associated with each species in the vulnerability factors 'displacement caused by structures' and 'displacement caused by vessels and/or helicopter activity'. The letters in Eqs. (1) and (2) represent the number of studies considered within each corresponding Method Category, and the numbers represent the Uncertainty Score associated with those Method Categories (Table 1). The different equations account for the different methods used to collect data pertaining to the four vulnerability factors (Table 1).

$$\text{Method Score} = (A)+(B \times 2)+(C \times 3)+(D \times 4)+(E \times 5) \quad (1)$$

$$\text{Method Score} = (A)+(B \times 2)+(C \times 3) \quad (2)$$

2.1.2. Combined Score

Combining the scores from each of the five criteria (Z: Species Score, Number of Sites, Number of Studies, Mean Years, Method Score) provided an estimation of uncertainty inherent in the data considered for each species in each vulnerability factor (Eq. (3)).

$$\text{Combined Score} = \sum_{i=Z}^{5} \text{score}_i \quad (3)$$

Combined Scores were assigned to one of five Uncertainty Levels and an associated Uncertainty Score (very low: 5, low: 4, moderate: 3, high: 2, and very high uncertainty: 1). To allocate Combined Scores to Uncertainty Levels, the greatest Combined Score for each vulnerability factor was rounded to the nearest ten and divided into five equal ranges to correspond to five Uncertainty Levels (Table 1). This provided a measure of uncertainty inherent in the data underlying species' vulnerability rankings in the four vulnerability factors (Supplementary Tables 2–5). For each species, the four Uncertainty Scores (generated from the Combined Score in each vulnerability factor) were summed to provide an overall estimation of uncertainty (Overall Uncertainty Score) associated with each species (Table 2).

In this paper, the term 'uncertainty' refers to the level of confidence in the data used to derive vulnerability rankings; based on the quality, quantity and relevance of that data. The Uncertainty Categories and Scores presented are generated based only on the data considered in this study (see Supplementary Material for data sources) and provide a relative estimation of uncertainty inherent in the data considered in each of the four vulnerability factors. Uncertainty Categories and Scores are measured on an ordinal scale; which means that the categories are ordered according to numerical values but that the numerical quantities represented by those values have no significance beyond allowing a ranking to be established. The Uncertainty Scores are labels that represent a categorical order and do not represent any concept of equal interval between categories. Uncertainty Categories and Scores do not represent an absolute scale and should not be taken to suggest that additional data collection may not be beneficial, even for those species associated with very low uncertainty. For example, if results indicate a very low uncertainty surrounding a particular species' flight altitude because of a large quantity of data available for that species, there may still be a poor understanding of the influence of different behaviour or weather conditions on flight height. As such, additional data collection could prove beneficial, as a better understanding of flight altitude would improve collision risk estimations.

2.2. Modification of vulnerability indices

2.2.1. Differing responses to structures and vessels and/or helicopters

Developments in understanding how seabirds respond to MREDs indicate that some species (e.g. Northern gannets *Morus bassanus*) react differently to structures (e.g. offshore wind turbines) than to vessels and helicopters. This study modifies methods presented in Furness et al. [3,4] to separately rank species according to vulnerability to i) structures, and ii) vessels and/or helicopters; rather than present a combined vulnerability factor. Greater weighting was applied to displacement/disturbance caused by structures (a) than to displacement/disturbance caused by vessels and/or helicopters (b) when calculating vulnerability to displacement/disturbance caused by offshore wind farms (Eq. (4)). This incorporates a likely greater influence of permanent structures over transient vessel and helicopter traffic. A measure of habitat specialisation (c) and a species conservation score (see [3,4]) were included (Eq. (4)).

$$\text{Displacement/disturbance score} = \frac{(((a \times c)+b) \times \text{conservation score})}{10} \quad (4)$$

2.2.2. Reduced risk of collision if displaced by structures

Birds avoiding and/or displaced by structures reduce their risk of collision. This study modifies the Furness et al. [3] calculation ranking seabird vulnerability to collision with offshore wind turbines by dividing the time spent at altitudes overlapping with turbine blades (d) by the level of displacement caused by structures (a) to incorporate this. Flight agility, percentage of time spent in flight, nocturnal flight activity (Y) and a species conservation score (see [3,4]) were included (Eq. (5)).

$$\text{Collision risk score} = \frac{d}{a} \times \frac{1}{3} \sum_{i=Y}^{3} \text{score}_i \times \text{conservation score} \quad (5)$$

All four vulnerability indices presented in Furness et al. [3,4] were recalculated following modification of index calculations and inclusion of new data (see Supplementary Tables 6–9 and Supplementary Reference List).

3. Results

3.1. Calculating uncertainty

There is greater uncertainty in our understanding of species' vulnerability to displacement caused by vessel and/or helicopter traffic, and seabird use of tidal races, than in data regarding the percentage of flight overlapping with wind turbine blades and the level of displacement caused by structures. Results indicate varying uncertainty among species in the four vulnerability factors, with storm petrels, sooty shearwater *Puffinus griseus* and Arctic skua *Stercorarius parasiticus* associated with very high uncertainty in three of the four vulnerability factors. Common goldeneye *Bucephala clangula*, greater scaup *Aythya marila*, long-tailed duck *Clangula hyemalis*, Manx shearwater *Puffinus puffinus*, roseate tern *Sterna dougallii*, white-tailed eagle *Haliaeetus albicilla* and grebes were associated with very high and high uncertainty (Table 2).

3.2. Modification of vulnerability indices

Overall seabird vulnerability rankings remained broadly the same, with only minor changes, following modification and recalculation of the four vulnerability indices presented in Furness et al. [3,4] (Supplementary Tables 6–9). Northern gannets

Table 2
Uncertainty inherent in data underlying the generation of four vulnerability factors for 38 seabird species. Uncertainty Scores equate to five Uncertainty Categories with greater scores indicating lower uncertainty: very high (score 1), high (score 2), moderate (score 3), low (score 4) and very low uncertainty (score 5). These categories and scores are on an ordinal scale where the numerical values have no significance beyond allowing a ranking to be established. Species rankings and scores were generated relative to data considered in each of the four vulnerability factors.

Species	Uncertainty Level: % of time at altitudes overlapping with turbine blades	Uncertainty Score	Uncertainty Level: Displacement caused by structures	Uncertainty Score	Uncertainty Level: Displacement caused by vessels and/or helicopters	Uncertainty Score	Uncertainty Level: Use of tidal races	Uncertainty Score	Overall Uncertainty Score (max 20)
European storm-petrel	Very high	1	Very high	1	High	2	Very high	1	5
Leach's storm-petrel	Very high	1	Very high	1	High	2	Very high	1	5
Sooty shearwater	Very high	1	Very high	1	High	2	Very high	1	5
Arctic skua	Moderate	3	Very high	1	Very high	1	Very high	1	6
Common goldeneye	Very high	1	Very high	1	High	2	High	2	6
Greater scaup	Very high	1	Very high	1	High	2	High	2	6
Manx shearwater	High	2	Very high	1	High	2	Very high	1	6
Slavonian grebe	Very high	1	High	2	High	2	Very high	1	6
White-tailed eagle	Very high	1	High	2	High	2	Very high	1	6
Great-crested grebe	High	2	High	2	High	2	Very high	1	7
Long-tailed duck	Very high	1	High	2	High	2	High	2	7
Roseate tern	Very high	1	High	2	High	2	High	2	7
Great skua	Moderate	3	High	2	High	2	Very high	1	8
Little tern	Very high	1	Moderate	3	Very high	1	Moderate	3	8
Velvet scoter	High	2	Very high	1	Moderate	3	High	2	8
Black-headed gull	Moderate	3	Moderate	3	High	2	Very high	1	9
Northern fulmar	Low	4	High	2	High	2	Very high	1	9
Arctic tern	Moderate	3	Moderate	3	High	2	High	2	10
Great northern diver	High	2	High	2	Very high	1	Very low	5	10
Little auk	Very high	1	Low	4	Low	4	Very high	1	10
Black-throated diver	High	2	Moderate	3	High	2	Low	4	11
Common gull	Low	4	Low	4	High	2	Very high	1	11
Common eider	Moderate	3	Moderate	3	Moderate	3	Moderate	3	12
Sandwich tern	Low	4	Low	4	High	2	High	2	12
Black guillemot	Very high	1	High	2	Very low	5	Very low	5	13
European shag	High	2	Low	4	High	2	Very low	5	13
Great black-backed gull	Low	4	Very low	5	Moderate	3	Very high	1	13
Great cormorant	Moderate	3	Very low	5	High	2	Moderate	3	13
Black-legged kittiwake	Very low	5	Very low	5	High	2	High	2	14
Common tern	Very low	5	Low	4	High	2	Moderate	3	14
Herring gull	Very low	5	Very low	5	Moderate	3	Very high	1	14
Lesser black-backed gull	Very low	5	Very low	5	Moderate	3	Very high	1	14
Northern gannet	Very low	5	Very low	5	High	2	High	2	14
Red-throated diver	Low	4	Low	4	High	2	Low	4	14
Common scoter	Low	4	Very low	5	Low	4	High	2	15
Atlantic puffin	Moderate	3	Moderate	3	Very low	5	Very low	5	16
Razorbill	Low	4	Very low	5	Very low	5	Low	4	18
Common guillemot	Low	4	Very low	5	Very low	5	Very low	5	19

increased in vulnerability to displacement by wind farms but decreased in vulnerability to collision with offshore wind turbines; cormorant species decreased in vulnerability to displacement but increased in vulnerability to collision; and auks species, including common guillemot *Uria aalge*, razorbill *Alca torda* and Atlantic puffin *Fratercula arctica*, were ranked as more vulnerable to displacement.

4. Discussion

Uncertainty associated with data used to calculate vulnerability indices is not always presented but is vital to make useful predictions. This study used Furness et al. [3,4] to develop novel methods to demonstrate uncertainty associated with vulnerability indices. This was achieved by assigning uncertainty to four measures of vulnerability for 38 Scottish seabird species to highlight where evidence supporting vulnerability rankings is more robust, where caution in interpreting results may be required, and where additional monitoring and research would be beneficial. The study also consolidates data from Furness et al. [3,4] with recent findings to consider developments in understanding how seabirds respond to MREDs and to present up-to-date vulnerability predictions.

4.1. Uncertainty indices

Being transparent and explicit about uncertainty is important to ensure consistent consideration of uncertainty inherent in vulnerability rankings. In assigning uncertainty to measures of vulnerability, this study identifies areas lacking data and highlights where caution in interpreting vulnerability index results should be adopted.

Results indicate greater uncertainty in data regarding displacement caused by vessels and/or helicopters, and use of tidal races by seabirds, than in data regarding the percentage of flight overlapping with wind turbine blades and the level of displacement caused by structures. This is because the offshore wind industry has developed more rapidly than other technologies [25] and establishing a level of collision and displacement caused by structures is a key component to gaining consent. As such, more data exist relating to these factors; particularly regarding seabird flight altitudes (Supplementary Tables 2–5). Results also indicate varying uncertainty associated with vulnerability rankings among species. For example, white-tailed eagles were ranked as the species most vulnerable to collision with wind turbines, whilst lesser black-backed gulls *Larus fuscus* were ranked as the second most vulnerable species (Supplementary Table 6). The uncertainty indices indicate that white-tailed eagles have a 'very high' level of uncertainty (score 1) associated with data informing the percentage of time spent overlapping with wind turbine blades, whilst lesser black-backed gull data are associated with a 'very low' level of uncertainty (score 5). These differing uncertainty levels are a result of varying data quality, quantity and relevance: with data on flight altitudes for white-tailed eagle originating from two studies undertaken at two terrestrial wind farms, compared with 35 studies undertaken at 28 different sites for lesser black-backed gull (Supplementary Table 2). This example indicates the importance of being explicit about uncertainty inherent in vulnerability indices to highlight where caution in interpreting rankings might be required and where estimates are more robust. Those areas highlighted as lacking in data would particularly benefit from additional monitoring and research to improve predictions of how seabirds may be affected by MREDs.

Species may lack data for several reasons: 1) they may be uncommon and rarely recorded; 2) they may be difficult to detect (e.g. small species like storm petrels); 3) they may be active during sea states incompatible with surveying (e.g. shearwaters in conditions above Beaufort sea state 4); or 4) they may be absent from MRED sites because they do not occur there (e.g. coastal species at offshore wind farms). For example, rare species associated with high uncertainty caused by a lack of observations may be highly vulnerable to potential impacts of MREDs because they come from small populations. Conversely, species absent from MRED sites could be associated with high uncertainty but may not be vulnerable to MREDs. It is important to distinguish why species might be associated with high uncertainty to ensure appropriate monitoring efforts.

4.2. Vulnerability indices

Species rankings remained broadly the same following revision and recalculation of the Furness et al. [3,4] vulnerability indices (Supplementary Tables 6–9). Recently available data (see Supplementary Reference List for sources) tended to support previous scores rather than alter them, which gives confidence in the approach and the broad rankings of species' vulnerabilities used.

In some cases, vulnerability rankings did alter. For example, vulnerability of Northern gannets to collision with wind turbines decreased (Supplementary Table 1). This is attributed to the modified calculation that separately scores vulnerability to i) structures and ii) vessel and helicopter traffic (Eq. (4)) rather than combining the two potential threats. The modification incorporates new evidence that some species respond differently to structures than to vessels and/or helicopters; for example, gannets are displaced by structures (therefore reducing their risk of collision) but show little response to vessels and helicopters [12,23,24].

For some species, predicted vulnerability to wind farms increased. European shags *Phalacrocorax aristotelis*, great cormorants *Phalacrocorax carbo* and some tern species increased in vulnerability to collision with wind turbines because of evidence indicating attraction to wind farms; potentially for foraging or roosting opportunities [10,11,23,24] (Supplementary Table 1). Common guillemots, razorbills and Atlantic puffins increased in vulnerability to displacement caused by wind farms, as recent evidence indicates auks are displaced by structures and vessels [12,23,24] (Supplementary Table 2). Gannets also increased in their vulnerability to displacement caused by wind farm structures but were not ranked as highly vulnerable to overall displacement caused by wind farms because of their large foraging ranges (Supplementary Table 7) and the comparably small area of habitat loss represented by a single wind farm. However, displacement caused by wind farms could prove a greater issue for gannets, and other species, if the cumulative effects of several installations throughout foraging ranges are considered. In this study, vulnerability indices could not take into consideration cumulative effects, or assess differences in seabird vulnerability to MREDs based on seasonality and life stage, but these issues should be borne in mind when applying the results of vulnerability indices, and should be considered at a site-specific level.

5. Conclusion

These uncertainty indices present vital information for the application of vulnerability indices ranking seabird vulnerability to MREDs. Uncertainty measures can inform MRED impact assessment processes by identifying species of potential concern that lack data, and contribute to identifying post-consent monitoring and strategic research priorities. The combined uncertainty and vulnerability indices could be employed to complement MRED site characterisation and inform sectoral plans by identifying areas supporting species that may be sensitive to MREDs. Given the

evolving understanding of species' responses to MREDs, these indices should be viewed as a work in progress and would benefit from regular consolidation with new information.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.marpol.2016.04.045.

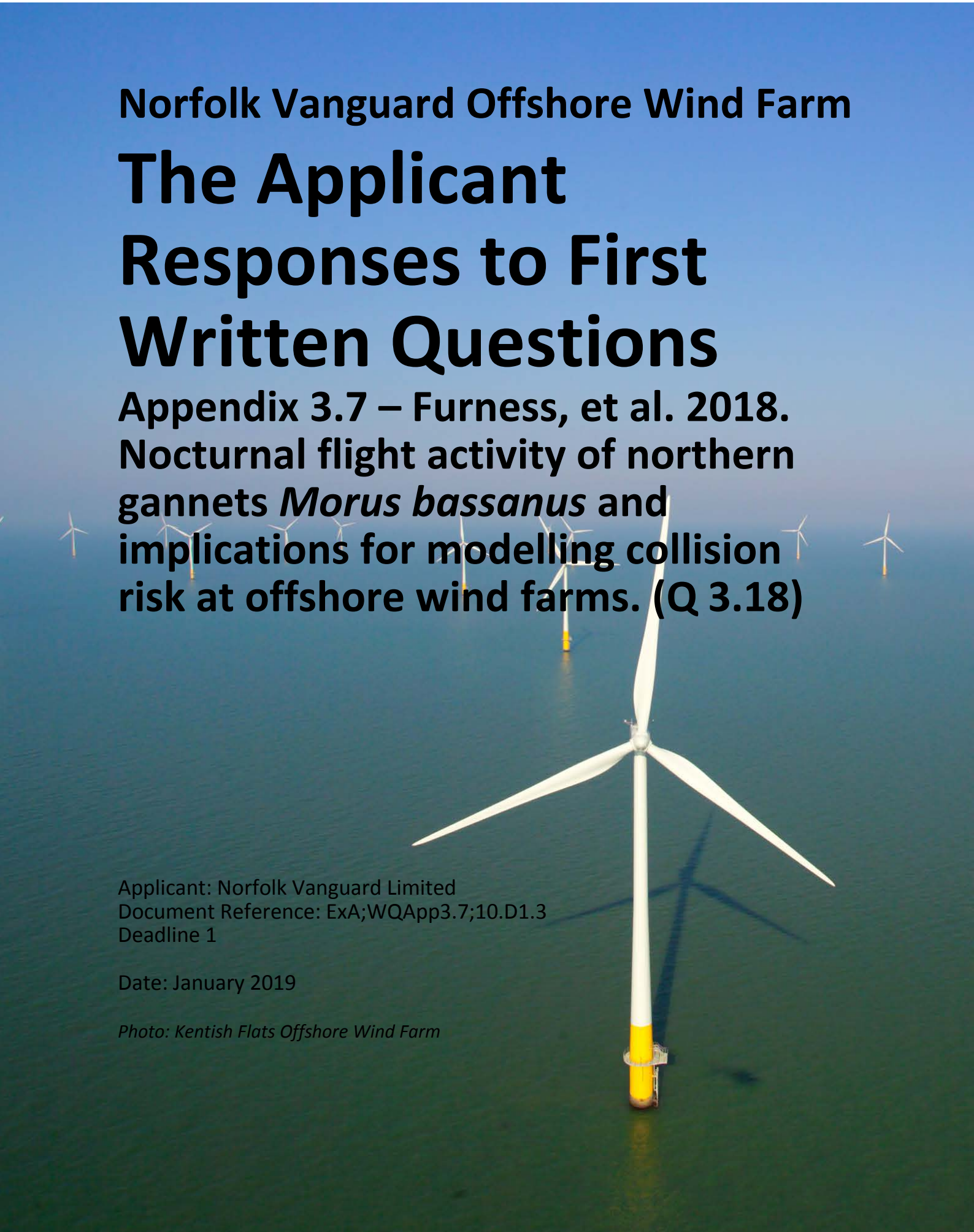
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Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.7 – Furness, et al. 2018. Nocturnal flight activity of northern gannets *Morus bassanus* and implications for modelling collision risk at offshore wind farms. (Q 3.18)



Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.7;10.D1.3
Deadline 1

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Photo: Kentish Flats Offshore Wind Farm

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Nocturnal flight activity of northern gannets *Morus bassanus* and implications for modelling collision risk at offshore wind farms

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ABSTRACT

Assessing the potential impacts of proposed offshore wind farm developments on seabird populations requires estimation of nocturnal flight activity of seabirds for input into collision risk models. One of the seabirds considered most at risk from collision with offshore wind turbines is the northern gannet *Morus bassanus*. The recommended correction for gannet nocturnal flight activity is currently a highly precautionary value. Here we use data from tracking studies to derive evidence-based correction factors for nocturnal flight activity of adult gannets during the breeding and nonbreeding seasons, and of immature gannets during the summer prospecting phase. Flight and diving activity of gannets was minimal during the night, astronomical and nautical twilight, for adults during the breeding season and nonbreeding season, and for immatures. Some flight activity occurred during the short period of civil twilight, but on average at about half the level seen during the day. Based on evidence from numerous tracking studies, we recommend that precautionary values of the nocturnal (sunset to sunrise) flight activity factor for estimating collision risk should be 8% of daytime flight activity during the breeding season and 3% of daytime flight activity during the nonbreeding season. Use of these evidence-based correction factors will improve the accuracy, and reduce the uncertainty of collision risk models, providing a more reliable assessment of the impacts of offshore wind farms on gannets.

1. Introduction

One of the key environmental issues facing developers of offshore wind farms in Environmental Impact Assessments is the impact that turbines may have on seabird populations as a consequence of mortality of birds that collide with rotating blades (Garthe and Hüppop, 2004; Furness et al., 2013). Bird collision mortality can be estimated using the Band collision risk model (Band, 2012). However, this requires an estimate of nocturnal flight activity as one of the model inputs. Seabird surveys at proposed offshore wind farm sites do not record the numbers of birds flying through the area at night, as visual (boat-based counts) or photographic (aerial) surveys are only practical during daylight hours. It is, therefore, necessary to use a correction factor, relative to daytime data, to allow for nocturnal flight activity of seabirds. Garthe and Hüppop (2004) assigned nocturnal flight activity scores to seabird species in five categories (scores of 1 to 5), based on existing limited evidence, their own judgement, and that of a panel of experts. They indicated that a score of 1 represented 'hardly any flight activity at night' while a score of 5 represented 'much flight activity at night'. These scores simply indicated that bird species that scored higher were likely to show more nocturnal flight activity than bird species that scored lower on the scale. Nevertheless Band (2012) advocated an arbitrary but precautionary translation of the Garthe and Hüppop (2004) scores for collision risk modelling as follows:

- 1 = 0% of daytime flight activity,
- 2 = 25% of daytime flight activity,
- 3 = 50% of daytime flight activity,
- 4 = 75% of daytime flight activity,
- 5 = 100% of daytime flight activity.

It is important to note that these suggested percentages were not based on evidence. It is also clear from Garthe and Hüppop (2004) that many of the scores for other seabird sensitivity metrics that they assigned were categorical rather than linear. Explicit examples are their scoring of population size; 1 ≥ 3million, 2 = 1–3 million,

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3 = 500,000–1 million, 4 = 100,000-500,000, 5 ≤ 100,000, and their scoring of flight altitude where scores 1 and 2 were based on median flight heights on a non-linear scale but scores 3 to 5 were based on 90th percentile flight heights on that scale.

One of the seabird species that appears most vulnerable to collision mortality at offshore wind farms is the northern gannet *Morus bassanus* (hereafter gannet) (Furness et al., 2013). The impact of collision mortality on gannet populations has been one of the primary concerns of recent planning applications for offshore wind farms. Band model calculations estimate a cumulative total of 2561 gannets per year may be killed by collisions at constructed and consented offshore wind farms in the United Kingdom sector of the North Sea (MacArthur Green, 2018). In relation to the Habitat Regulations Assessment (HRA) component of the planning application for Hornsea Two offshore wind farm, Natural England (2015) were unable to conclude beyond all reasonable scientific doubt that the estimated cumulative collision total for offshore wind farms would not have an adverse effect on the integrity of the Flamborough and Filey Coast proposed Special Protection Area (FFC pSPA) gannet population. In relation to East Anglia THREE offshore wind farm, The Planning Inspectorate (2017) stated "two key HRA matters were the focus of the Examination: The effect of the proposed development in combination with other offshore wind farms on the kittiwake and gannet features of the FFC pSPA". Therefore, estimated collision mortality of gannets has the potential to stop the considerable further development of offshore wind farms planned for the North Sea (The Crown Estate, 2018).

Garthe and Hüppop (2004) assigned a nocturnal flight activity score of two for gannets, based on evidence from Garthe et al. (1999, 2000, 2003) and Hamer et al. (2000), and this was converted to 25% of the daytime level by Band (2012). During mid-summer, the correction for nocturnal flight activity makes only a small difference to estimated numbers of collisions, since the night is short in mid-summer (Fig. 1). However, in winter the effect is larger: because the night is about twice as long as day during winter, the Band (2012) model estimates an additional 0.5 collisions at night for each collision during the day for an offshore wind farm located in the southern North Sea. Most offshore wind farms in Europe are in the southern North Sea. Gannet numbers in that region are low in summer and peak strongly during November (Stone et al., 1995; Furness et al., 2018), so the influence of nocturnal correction is likely to be close to the 0.5 nocturnal collisions per daytime collision. This means that an evidence-based correction for this parameter would be important in improving confidence in the estimated cumulative impact of collisions at offshore wind farms on gannet populations, especially where the cumulative total is close to a level that could result in consenting risk for further offshore wind farm developments.

Garthe and Hüppop (2004) did not provide an explicit definition of day and night. Collision risk modelling using the Band model defines day as sunrise to sunset and night as sunset to sunrise, with the estimation of the times of sunrise and sunset derived from Forsythe et al. (1995). However, that definition of night contrasts with the official concept of 'twilight' and 'night'. 'Civil twilight' is defined as from sunset to the sun falling 6° below the horizon and in the morning from when the sun reaches 6° below the horizon until sunrise. 'Nautical twilight' is defined as the sun being between 6° and 12° below the horizon. 'Astronomical twilight' is defined as the sun being between 12° and 18° below the horizon, and 'night' is from then until the sun has risen back to 18° below the horizon. In the regions where gannets overwinter, the transition through twilight can be rapid. However, in summer, there may be no official 'night' at all, because astronomical twilight persists if the sun never falls more than 18° below the horizon (Fig. 1). This suggests that a more subtle definition of 'day' and 'night' is required than that used in Band (2012) to take account of the considerable variation in light levels between sunset and sunrise in summer, and especially at higher latitudes. Since gannets are visual predators (Garthe et al., 2000, 2003; Lewis et al., 2002), it is likely that flight activity is determined by the minimum light levels to allow foraging, commuting or migrating. Cleasby et al. (2015a) noted that gannet dives tend to be shallower close to sunrise and sunset, which supports the argument that diving at twilight is limited by the birds' ability to see their prey.

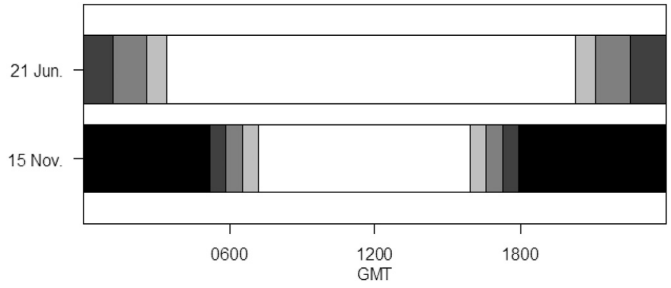


Fig. 1. Illustration of the differences in duration of night (black), astronomical twilight (dark grey), nautical twilight (medium grey), civil twilight (light grey), and day (white) at midsummer (21 June) and during peak migration of gannets through the southern North Sea (November) at 52°30′N 2°30′E, a typical location for a southern North Sea offshore wind farm. Times in GMT.

There are now many data sets showing flight activity levels of gannets at different times of day, both for breeding birds and for birds during the migration period and in winter. In this paper we assess the available evidence in order to provide evidence-based corrections for nocturnal flight activity of gannets for use in Band model collision assessments. This will give more accurate results than estimates based on the conversion of scores assigned by Garthe and Hüppop (2004). Here we consider data from throughout the range of the gannet. However, we focus on deriving appropriate corrections for use in examining impacts on gannets in the North Sea, the region with by far the largest number of constructed and proposed offshore wind farms (The Crown Estate, 2018).

2. Methods

We carried out a literature search, focused on Web of Knowledge and Google Scholar but also searching 'grey literature' (such as consultant reports and SNCB guidance documents) to find data on daytime and nocturnal flight activity of gannets. GLS logger data (from Garthe et al., 2012) were used in order to identify variation in flight behaviour according to the time of day. We considered activity data divided into 'day', 'civil twilight', 'nautical twilight', 'astronomical twilight', and 'night' (Fig. 1). We used Time and Date (2018) to extract timings of sunrise, sunset, civil, nautical, astronomical twilight and night appropriate for the location and date of each study.

We considered data derived from tags deployments: a) data from breeding gannets incorporated into Garthe and Hüppop (2004), b) data from breeding gannets collected since Garthe and Hüppop (2004), c) geolocator (GLS) data from gannets during the non-breeding season, d) data from tags on immature gannets (Jeglinski et al. unpublished data). Flight activity is frequently referred to as the percentage of each hour spent in flight. Several different types of tag have been used to infer at sea behaviours of seabirds. Travel speed of birds at sea derived from GPS tracking can be assigned to resting on the sea or to flying if there is a clearly bimodal distribution of travel speeds, with the faster mode representing flight (Grémillet et al., 2004). Few studies have used accelerometer data from tags, but these can aid interpretation of behaviour of birds (Warwick-Evans et al., 2015). Geolocator tags that have a salt-water switch provide accurate data for gannets because birds are either in the water (switch on) or flying (switch off), as it can be assumed that gannets are not on land during the nonbreeding season when away from the colony (Garthe et al., 2012). Some loggers record diving activity but not flight activity. Since gannets only dive from the air, and not from the sea surface, diving activity implies flight activity.

Table 1
Summary of flight and plunge-diving activity between sunset and sunrise in relation to daytime levels.

Reference	Season	Colony	Number of tags and tag type	Hours of data on foraging trips	Flight activity from sunset to sunrise as % of flight activity during day	Number of plunge dives recorded	% of plunge dives occurring between sunset and sunrise
Garthe et al., 1999	Breeding	Hermaness	3 stomach temperature	215	(20.9%)	32	(0%)
Garthe et al., 2000	Breeding	Funk Island	11 depth & wing-beat	–	–	336	4.3%
Hamer et al., 2000	Breeding	Bass Rock	17 Argos satellite PTTs	> 2000	0%	–	0
Garthe et al., 2003	Breeding	Funk Island	16 depth & temperature	330	6%	315	0.5%
Hamer et al., 2007	Breeding	Bass Rock	53 Argos PTTs, or GPS tags	> 5000	0%	–	–
Langston et al., 2013	Breeding	Bempton	42 Argos satellite PTTs	–	0%	–	–
Garthe et al., 2014	Breeding	Bonaventure	5–7 GPS & altimeter	–	–	731	15%
Warwick-Evans et al., 2015	Breeding	Alderney	9 GPS & accelerometer	678	–	1236	0%
Warwick-Evans et al., 2017	Breeding	Alderney	9 GPS & accelerometer	678	15.7%	–	–
Garthe et al., 2017	Breeding	Helgoland	14 GPS & depth	–	–	2557	0.7%
Garthe et al., 2012	Autumn	North Sea	59 geolocator & temperature	–	2.5%	–	–
Garthe et al., 2012	Winter	North Sea to Africa	50 geolocator & temperature	–	1.9%	–	–
This analysis	Autumn	Remaining in North Sea	4 geolocator & temperature	35 days	1%	–	–
This analysis	Autumn	Migrating to Africa	8 geolocator & temperature	71 days	3.8%	–	–
This analysis	Breeding	Bass Rock, immatures	7 geolocator & temperature	4 days	Very low	–	–
This analysis	Winter	Birds in North Sea	4 geolocator & temperature	–	2.4%	–	–

Details of tag types deployed in different studies and numbers of data sets obtained are summarised in Table 1. Garthe et al. (1999) deployed GPS loggers on adult gannets at Hermaness, Shetland. Their loggers provided data on feeding events and on flight activity throughout the 24-h period, but the study was limited to a sample of just three individuals tracked for a few days in mid-July 1997. Garthe et al. (2000) deployed time-depth loggers on adult gannets at Funk Island, Canada. Hamer et al. (2000) deployed satellite PTTs on chick-rearing gannets at the Bass Rock, Scotland. Garthe et al. (2003) deployed loggers on chick-rearing adult gannets at Funk Island, Canada recording diving activity and flight activity. Hamer et al. (2007) presented data from satellite tracking or GPS loggers over three breeding seasons for a total of 53 gannets breeding on the Bass Rock. RSPB deployed satellite PTTs and GPS tags on gannets breeding at Bempton during three breeding seasons (2010, 2011 and 2012), obtaining tracks of chick-rearing birds and some tracks of post-breeding dispersal and migration. Garthe et al. (2014) reported on diving activity of breeding birds from Bonaventure, Canada. Warwick-Evans et al. (2015) presented data on the diurnal pattern of plunge dives by gannets breeding at Alderney, Channel Islands. Garthe et al. (2017) reported on plunge diving activity of breeding birds from Helgoland, Germany. Garthe et al. (2012) deployed geolocator loggers on breeding adult gannets on the Bass Rock in 2002, 2003, and 2008, and presented data on flight and resting behaviour of those birds.

Although we were able to access published data on flight activity of adult gannets, no data on flight activity of immature gannets have been published. We therefore include new data on immature gannets. Jeglinski (unpublished data) deployed 7 GPS GSM tags (Pathtrack Ltd. Leeds, UK) on 2–3 year old immature gannets on the Bass Rock between the 3rd and the 11th of July 2016. These tags were equipped with a dynamic algorithm that adjusts the GPS fix rate to the battery voltage, and the GPS fix rate was programmed to 5 min intervals but in reality GPS fixes were taken every 10 ± 5.4 min. Based on the location and timestamp of each position, the data were categorized into four categories (dawn, day, dusk, night) using the function crepuscule (R package maptools). The function uses algorithms provided by the National Oceanic & Atmospheric Administration (NOAA) to implement flexibility for various formal definitions of times of dawn and dusk. The definition of dawn and dusk was based on a solar angle of $< 6^{o}$ below the horizon so represents civil twilight. The speed between successive GPS locations was calculated for each individual and each period. Data were divided into two categories based on the thresholds defined by Bennison et al. (2017) and Wakefield et al. (2013): < 3.5 km/h (likely corresponds to resting e.g. drifting on the sea surface), > 3.5 km/h which may indicate some flight activity as this speed is unlikely to occur due to drift alone, although erroneous high speeds can occasionally occur as a result of inaccuracies in GPS logger location estimates (S. Garthe unpubl. data). We overlay the GPS locations at night with an ocean shapefile based on ocean coastlines at a scale of 10 m (Natural Earth, 2017) to identify if immatures spend the night at sea or on land. All locations were positioned at sea, so we concluded that immatures do not sleep on land, which is a pre-requisite for assigning behaviours as either flying or resting on the sea.

Although differences in methodology among studies might make comparisons of activity budgets between studies difficult, in this paper we only make comparisons within studies, comparing between periods of the day, and so the methodology used in defining flight activity is identical between the relevant periods of the day that we compare.

3. Results

3.1. Breeding adults

Garthe et al. (1999, 2000, 2003, 2014), Hamer et al. (2000, 2007), and Warwick-Evans et al. (2015, 2017) reported that breeding gannet flight activity was negligible during nautical twilight, astronomical

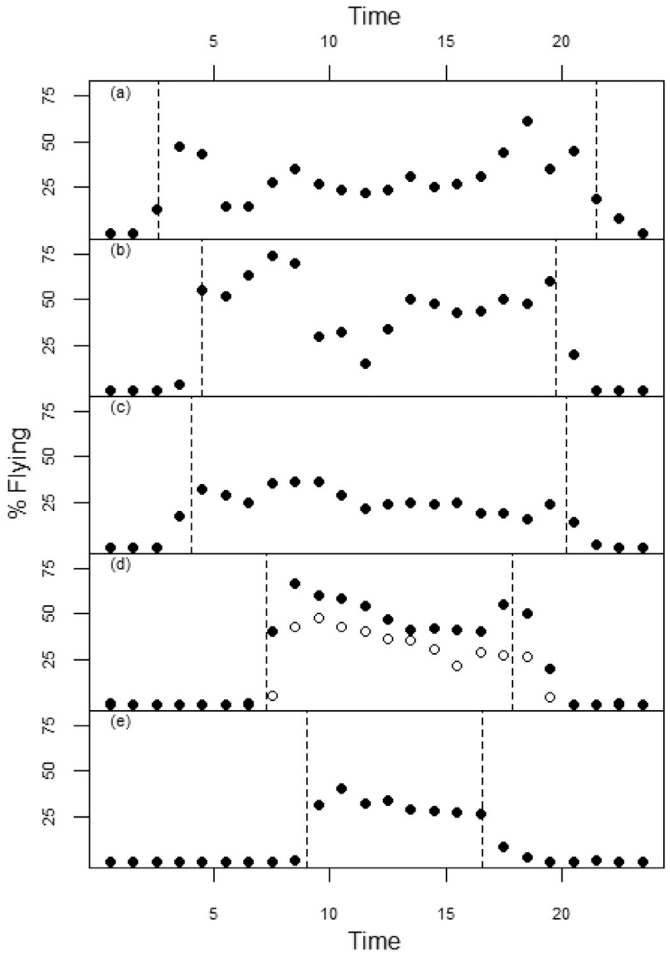


Fig. 2. Flight activity of gannets in relation to hour of the day a) in mid-July 1997 at Hermaness, Shetland (Garthe et al., 1999); b) of chick-rearing gannets from Funk Island, Canada (Garthe et al., 2003); c) of breeding adult gannets at Alderney, Channel Islands in June 2013 (Warwick-Evans et al., 2017); d) in autumn migration (mid-late October) of adult gannets that overwinter in the North Sea (open symbols) or were migrating to overwinter in west Africa (solid symbols) (Garthe et al., 2012); e) in December by adult gannets from Bass Rock that remained in the southern North Sea through winter (Garthe et al., 2012). Dashed lines show times of sunrise and sunset.

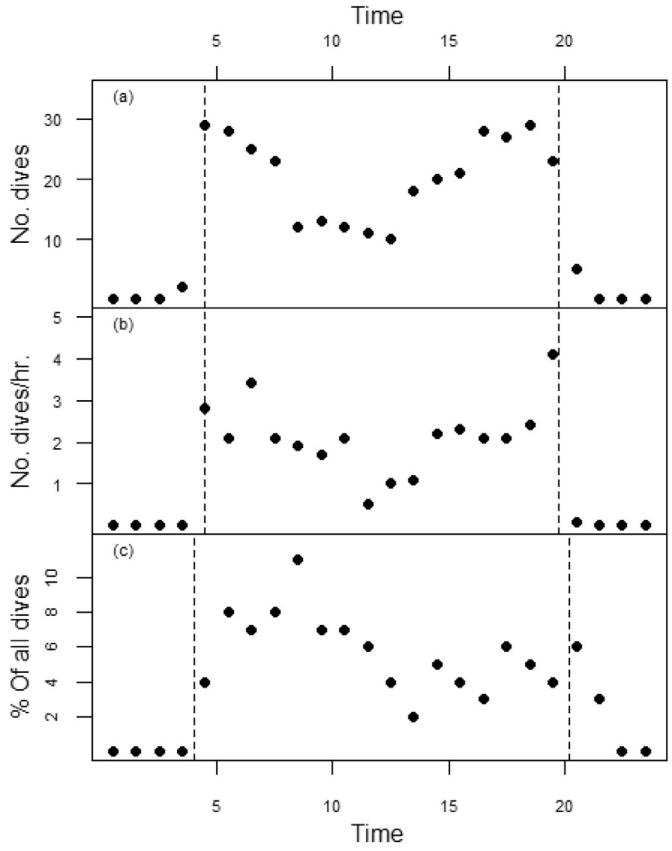


Fig. 3. Diving activity of chick-rearing gannets from a) Funk Island, Canada at different times of day (Garthe et al. (2000)). b) Funk Island, Canada at different times of day (Garthe et al. (2003)). c) Bonaventure Island, Canada at different times of day (Garthe et al. (2014)). Dashed lines show times of sunrise and sunset.

twilight and night, and was much lower during civil twilight than during the day. This was also inferred by Langston et al. (2013). The numerous studies of breeding gannets at different colonies and deploying a range of different types of tag show consistent results. Flight (Fig. 2a-e) and diving activity (Fig. 3a-c) of gannets occurred throughout the daylight period, sometimes with a slight tendency to peak just after sunrise and to a lesser extent just before sunset. Flight and diving activity were lower immediately after sunset and immediately before sunrise than during the day, and fell to negligible levels shortly after sunset. Flight activity remained at negligible levels through the night until shortly before sunrise (Fig. 2a-e). When averaged over the period from sunset to sunrise, flight activity of breeding adults averaged 7.1% of the daytime level across six studies, while diving by breeding adults averaged 2.9% of the daytime level across seven studies (Table 1).

3.2. Immatures

Tracked immature gannets moved at an average speed of 1.4 ± 1.21 (SE) km/h during nautical and astronomical twilight and night. Based on the threshold metric, 95.3% of the time periods were defined as birds resting on the water (Electronic Supplement Table S1). During dawn the average speed was 3.67 ± 6.44 km/h, which is higher than at night, so suggests some flight activity. During dusk the average speed was 8.87 ± 14.81 km/h, also higher than at night and suggesting some flight activity. Based on the threshold metric, on average 83.3% and 65.8% of dawn and dusk periods were identified as birds resting on the water. With a typical flight speed of 25 km/h (e.g. Hamer et al., 2000), birds would have to spend < 25% of a period in flight even during the few periods when flight appears likely, in order to result in a mean speed of movement of only 4 to 5 km/h, which was the average maximum speed recorded across the sample of birds during night in the few intervals when some flight activity was indicated (Electronic Supplement Table S2). During the day, 61.7% of time intervals were classified as birds resting on the sea surface. However, during the day, immatures moved with an average speed of 11.48 ± 17.14 km/h, reaching maximum speeds between 55 and 87 km/h.

Even though the dominant activity during night was resting, all birds apparently spend a very small proportion of the night in flight. The data show that immature flight activity, based on flight speed, was higher during civil twilight than at night, with birds generally flying faster (and spending almost double the proportion of time periods during which there seemed to be some flight activity) during dusk than during dawn. Overall, the data show low flight activity by immatures between sunset and sunrise, though possibly slightly more than seen among breeding adults.

3.3. Adults in the non-breeding season

Flight activity of adult gannets during the non-breeding season between sunset and sunrise averaged 2.5% of daytime level during autumn and 1.9% of daytime level during winter (Garthe et al., 2012). Data for the subset of birds migrating through the North Sea in autumn, which may be the most appropriate in relation to collision risk at North Sea offshore wind farms, are presented in Fig. 2d. During peak autumn migration (mid-late October) adult gannets from the Bass Rock that were going to remain in the North Sea or Channel overwinter spent on average 31.9% of daylight time in flight, 2% of civil and nautical twilight combined in flight and 0% of astronomical twilight and night time in flight. Considering the nocturnal period (sunset to sunrise) flight activity of adult gannets in the North Sea that remained in the area overwinter averaged 1% of the daytime level, whereas flight activity of adult gannets that migrated to winter off southern Europe or west Africa averaged 3.8% of the daytime level (Table 1).

Data for flight activity during winter of birds remaining in the southern North Sea during winter are presented in Fig. 2e. In winter (December), birds spent hardly any of the night or astronomical twilight in flight, with a mean of 0.2% of the night spent flying. Garthe et al. (2012) showed that during daylight hours, birds spent more time flying in autumn than in winter, and birds that were migrating to West Africa spent more time flying during the day (40% of daylight hours) than birds that wintered in UK waters (30% of daylight hours flying). In winter, birds spent on average 26% of daylight hours in flight (Garthe et al., 2012). However, birds wintering in the southern North Sea flew more than birds wintering in west Africa, so that the daytime baseline level of flight activity against which nocturnal activity is compared needs to be considered on a regional basis to ensure a like-for-like comparison. Birds in the southern North Sea in December flew 31% of daylight hours. By comparison, they flew 4.5% of civil twilight and nautical twilight periods, and 0.2% of astronomical twilight and night periods. Flight activity between sunset and sunrise averaged 0.75% of the time, compared with 31% of the time between sunrise and sunset. Flight between sunset and sunrise therefore averaged 2.4% of the rate during the day (Table 1).

4. Discussion

Multiple studies of breeding gannets in multiple years at colonies in Scotland, Germany, Canada and the Channel Islands, and one of immature gannets, all show extremely low levels of flight activity of gannets at night, and no plunge diving at night (Table 1). Data from birds in the non-breeding season show flight activity between sunset and sunrise is consistently around 1% to 4% of the amount recorded during daytime. Data from breeding adults show flight activity between sunset and sunrise averaging 7% of the daytime level, with diving activity between sunset and sunrise averaging 3% of the daytime level (Table 1). Higher nocturnal flight activity of breeding adults than of adults during the non-breeding season could possibly reflect high energy demands of breeding, forcing birds to extend foraging effort, but may simply result from the fact that almost all nocturnal flight activity occurs during civil twilight. In summer, civil twilight lasts longer than in winter (Fig. 1), and there may be no astronomical twilight or night. In winter, night represents a much greater proportion of the period from sunset to sunrise (Fig. 1), so lack of flight activity at night reduces the average level of flight activity between sunset and sunrise during winter compared to summer.

The logger data from non-breeding adult gannets are robust as they are from a large sample size over several winters (Garthe et al., 2012), and can be disaggregated by region and by migration extent of individuals. The low level of flight activity at night is consistent with the understanding of gannet natural history; as visual hunters gannets will not be able to locate fish on which to plunge-dive during hours of darkness (Lewis et al., 2002; Cleasby et al., 2015a), and in the non-breeding season will not need to fly at night to return to nest sites. Gannet migrations are very slow compared to migrations of other seabird species (Garthe et al., 2012; Fifield et al., 2014) and so birds are not under any pressures to migrate during the night.

It is unclear whether differences in estimates of nocturnal flight activity among studies of breeding adults represent differences in behaviour of birds from different colonies or simply reflect chance variation. The largest estimates of the amount of nocturnal flight activity by breeding adults tended to come from the studies based on the smallest sample sizes; a weighted average based on the number of birds in each study would reduce this estimate by about 50%. However, it is also possible that ecological conditions affect amount of flight activity by breeding gannets. The highest estimate of flight activity between sunset and sunrise (20.9% of the daytime level) was from Garthe et al. (1999) who studied just three birds at a colony in Shetland. At that latitude in summer there is a long period of civil twilight, and no astronomical twilight or night. Birds at higher latitude colonies (such as Shetland) might show relatively more nocturnal flight because civil twilight represents a greater part of the period from sunset to sunrise than further south.

Diving activity (and therefore foraging rather than commuting flight) was even less frequent between sunset and sunrise than flight activity. Given that collision risk is higher when gannets are foraging rather than when they are commuting (Cleasby et al., 2015b), the low amount of foraging flight during the twilight period will further reduce collision risk at that time of day compared to flight during the day. This suggests that a case could also be made for using a lower flight height distribution for the few birds still flying during twilight compared with that used for gannets flying during the day.

Gannet flight activity differs considerably between daytime, civil twilight, and darker periods (nautical twilight, astronomical twilight and night). Thus there would be merit in developing a more nuanced Band model taking account of the activity patterns in these different periods. This, additionally, would account for the very different durations of twilight and night at different times of year. However, in the short term, predictions from Band modelling could be improved by adopting the evidence-based values for flight activity during the nocturnal period (i.e. from sunset to sunrise) in the current Band model.

Based on the average percentage of daytime flight activity that was observed between sunset and sunrise, we recommend that precautionary values of the nocturnal activity factor used with the Band model for estimating collisions should be 8% during the breeding period and 3% during the nonbreeding period. This would not require the Band model to be altered to add separate calculations for twilight periods, although that might be a longer term objective. These values are strongly founded on evidence, and are more appropriate than the 25% value currently suggested by Band (2012) which was not evidence-based. Furthermore, we consider that these evidence-based estimates remain precautionary because they use the unweighted average across studies, and a weighted average accounting for sample sizes would reduce the estimate for breeding birds further. Tracking data exist for several other seabird species. However, most data sets have not been published in a form that allows nocturnal flight activity to be seen. Given that the evidence-based estimates for gannet represent a large reduction from the value employed by Band (2012), there would be merit in analysing nocturnal flight activity of species such as kittiwake *Rissa tridactyla*, great black-backed gull *Larus marinus* and lesser black-backed gull *L. fuscus*. Evidence-based estimates for those species would also help to reduce uncertainty in environmental impact assessments for offshore wind farms.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2018.06.006.

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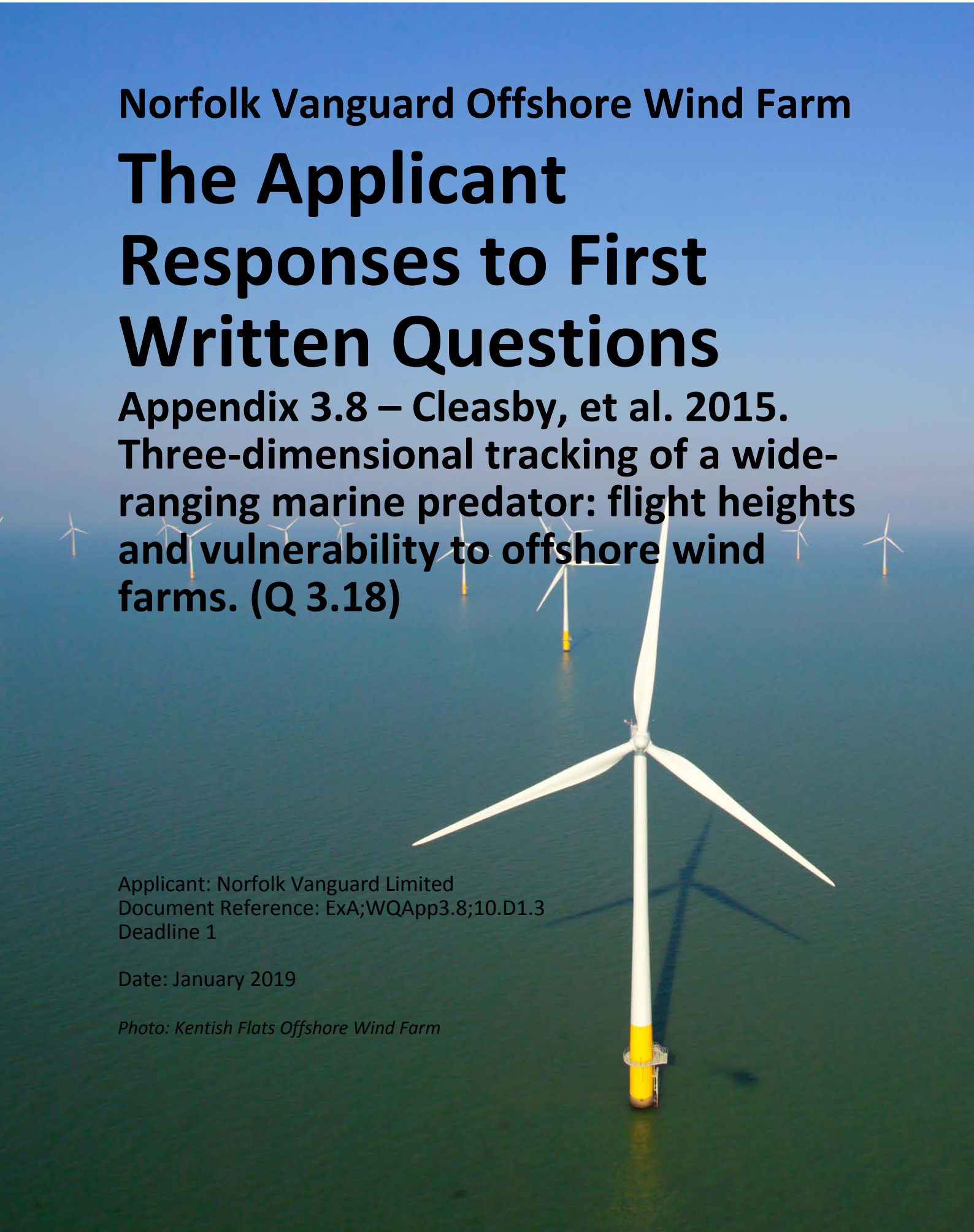
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The Applicant Responses to First Written Questions

Appendix 3.8 – Cleasby, et al. 2015. Three-dimensional tracking of a wide-ranging marine predator: flight heights and vulnerability to offshore wind farms. (Q 3.18)



Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.8;10.D1.3
Deadline 1

Date: January 2019

Photo: Kentish Flats Offshore Wind Farm

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Three-dimensional tracking of a wide-ranging marine predator: flight heights and vulnerability to offshore wind farms

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Abstract

1. A large increase in offshore wind turbine capacity is anticipated within the next decade, raising concerns about possible adverse impacts on birds as a result of collision risk. Birds' flight heights greatly influence this risk, yet height estimates are currently available only using methods such as radar- or ship-based observations over limited areas.

2. Bird-borne data-loggers have the potential to provide improved estimates of collision risk and here, we used data from Global Position System (GPS)-loggers and barometric pressure loggers to track the three-dimensional movements of northern gannets rearing chicks at a large colony in south-east Scotland (Bass Rock), located < 50 km from several major wind farm developments with recent planning consent. We estimated the foraging ranges and densities of birds at sea, their flight heights during different activities and the spatial variation in height during trips. We then used these data in collision-risk models to explore how the use of different methods to determine flight height affects the predicted risk of birds colliding with turbines.

3. Gannets foraged in and around planned wind farm sites. The probability of flying at collision-risk height was low during commuting between colonies and foraging areas (median height 12 m) but was greater during periods of active foraging (median height 27 m), and we estimated that ~1500 breeding adults from Bass Rock could be killed by collision with wind turbines at two planned sites in the Firth of Forth region each year. This is up to 12 times greater than the potential mortality predicted using other available flight-height estimates.

4. *Synthesis and applications*. The use of conventional flight-height estimation techniques resulted in large underestimates of the numbers of birds at risk of colliding with wind turbines. Hence, we recommend using GPS and barometric tracking to derive activity-specific and spatially explicit flight heights and collision risks. Our predictions of potential mortality approached levels at which long-term population viability could be threatened, highlighting a need for further data to refine estimates of collision risks and sustainable mortality thresholds. We also advocate raising the minimum permitted clearance of turbine blades at sites with high potential collision risk from 22 to 30 m above sea level.

Key-words: climate change, collision risk, environmental impact, foraging, *Morus bassanus*, offshore renewables, population model, wildlife telemetry

Introduction

The number of offshore wind turbines is predicted to increase tenfold over the next decade, particularly in the shallow seas of Europe, making predicting the likely ecological impacts of these developments a conservation priority (Infield 2013). The impact of offshore wind farms on the internationally important populations of seabirds that forage in the seas around the UK is of particular concern (Masden *et al.* 2012; Furness, Wade & Masden

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2013). Evidence from both onshore and offshore wind farms shows that, for some birds, both direct mortality due to collisions and indirect disturbance could have significant negative impacts (Marques *et al*. 2014). However, estimates of mortality from collisions with turbines are frequently based on partial or subjective data (Ferrer *et al*. 2012). Moreover, accurate data on spatial, temporal and activity-specific variability in flight heights are lacking, especially for seabirds (Johnston *et al*. 2014).

To date, seabird flight heights have largely been estimated using ship-based surveys involving surveyors' subjective assessment by eye, the accuracy of which remains unquantified (Cook *et al*. 2012; Johnston *et al*. 2014). Radar can quantify flight heights and tracks much more accurately (Desholm *et al*. 2006), but provides data over only a limited area (maximum range typically 6 km) and is costly. An additional limitation of both ship-based surveys and radar is that the provenance, breeding status and in many cases age and sex of birds cannot be determined, yet these parameters are needed to model population-level consequences of predicted mortality more accurately.

A recent review of the vulnerability of marine bird populations to wind farms identified northern gannets *Morus bassanus* (Linn.) as a species of particular concern (Furness, Wade & Masden 2013). Britain and Ireland hold ~65% and 85% of the world and European breeding populations of gannets, respectively (Wanless, Murray & Harris 2005; Murray, Harris & Wanless 2015). Planning consent for several major offshore wind farms has just been granted in close proximity to major North Sea Gannet colonies (Furness & Wanless 2014). Tracking studies suggest that gannets typically travel at heights below 10 m but may ascend to above 20 m during foraging, as they search for prey and attain height, and therefore potential energy, prior to plunge-diving (Garthe *et al*. 2014). Foraging gannets may thus be at significant risk of colliding with wind turbine blades, which may sweep an area from 22 m above sea level at highest astronomical tide (the lowest blade clearance currently permitted; UK Maritime and Coastguard Agency 2008) to around 160 m above sea level or higher (Rothery, Newton & Little 2009; Furness, Wade & Masden 2013). However, collision risk is currently difficult to estimate because accurate data on flight heights and behaviour in proposed wind farm sites are lacking.

Global Position System (GPS) loggers are increasingly used to record the movements of birds (e.g. Wakefield *et al*. 2013), and can provide information on height as well as latitude and longitude. However, whilst they are accurate to <10 m in the horizontal plane, vertical measurement errors may be an order of magnitude greater (Ladetto *et al*. 2000). Alternatively, height may be estimated by recording atmospheric pressure and comparing it to that at sea level using the barometric formula (Berberan-Santos, Bodunov & Pogliani 1997; Wallace & Hobbs 2006). For example, the flight heights of tropical seabirds have been estimated by assuming that atmospheric pressure recorded at a reference location, such as a bird's breeding colony, is representative of that at tracking locations (Weimerskirch *et al*. 2005). Errors incurred using this approach are tolerably small in tropical regions, where atmospheric pressure usually varies little over wide areas, but at temperate latitudes, surface atmospheric pressure is much more labile in time and space, potentially introducing large errors. To obtain more accurate estimates of flight height in these areas, pressure loggers must therefore be calibrated frequently against pressure at sea level in their vicinity (Li, Harvey & Gallagher 2013). The reliability of such barometric altitude estimates can be assessed by examining the correlation with GPS altitude because, whilst individual GPS altitudes are unreliable, errors have a Gaussian distribution with a mean of zero (Whang & Ra 2008). Hence, a reliable pressure-logger-based estimate should correlate with GPS-derived altitude.

The Bass Rock (56°6′N, 2°36′W) supports the world's largest breeding population of gannets (~75 000 breeding pairs in 2014; Murray, Harris & Wanless 2015), yet several large wind farms have been scheduled for construction in its vicinity within the next 5 years (Marine Scotland 2014; Fig. 1). Taking this system as an example of one in which the potential effects of offshore wind farms on seabirds are of concern, we aimed to assess how the use of different flight-height estimation methods affects the predicted risk of birds colliding with turbines and hence the potential impact on seabird population viability. We combined GPS tracks of gannets breeding at the Bass Rock in three consecutive years (2010–2012) with data from bird-borne pressure loggers in 2011 and 2012 to estimate the foraging ranges and densities of birds at sea, their flight heights during different activities, and spatial variation in flight height and potential collision risk during foraging trips. We then compared these results to collision risks predicted using flight heights reported from ship-based and radar-based studies.

Materials and methods

STUDY SITE AND SAMPLING

Fieldwork took place at Bass Rock between mid-June and mid-August over three consecutive breeding seasons (2010–2012). Adult gannets raising chicks (n = 49 in 2010, 25 in 2011, 33 in 2012) were caught at the nest using a 6-m telescopic pole fitted with a wire crook (n = 55 birds in total; most were sampled in >1 year). Upon initial capture, birds were fitted with a metal British Trust for Ornithology ring and an individually numbered plastic colour-ring (Wakefield *et al*. 2013). An i-gotU 200/600 GPS logger (Mobile Action Technology, Taipei, Taiwan) weighing ~30 g was attached to the upper side of the central three tail feathers of each bird using Tesa© tape (Milton Keynes, UK). In addition, a subset of these birds caught in 2011 (n = 11) or 2012 (n = 5) was equipped with a pressure logger (MSR-145; MSR Electronics, Seuzach, Switzerland) weighing 18 g and taped to the underside of the central tail feather. Air pressure and temperature were logged at 1 Hz, which resulted in a pressure logger battery life of ≥7 days. GPS-loggers, programmed to record at either high

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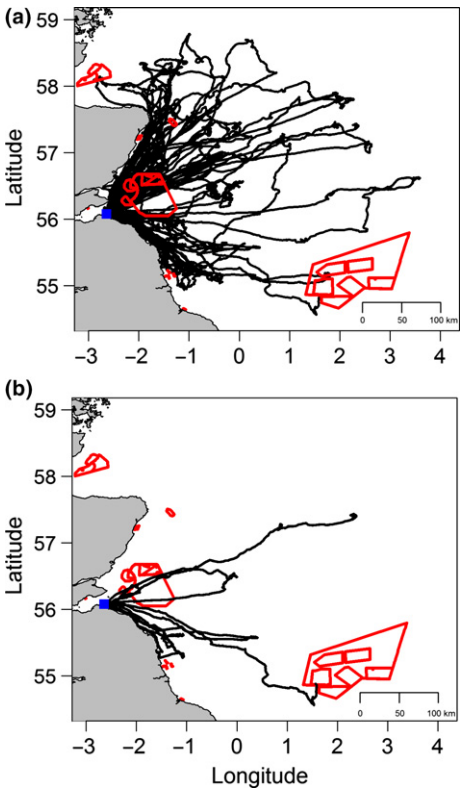


Fig. 1. Foraging tracks of chick-provisioning northern gannets from Bass Rock during June–August in 2010–2012 (black) and locations of proposed wind farms: (a) all tracks for birds equipped with Global Position System-loggers and altitude loggers (n = 54 trips by 16 birds); (b) tracks recorded at high resolution (1 Hz; n = 7 trips by 7 birds). For details of wind farm sites, see Marine Scotland (2014). *S*quare shows location of Bass Rock.

resolution (1 Hz; i.e. once per second) or standard resolution (2-min intervals), were deployed along with pressure loggers. We collected high-resolution GPS data to discriminate behaviours over very short time intervals, allowing us to assess flight heights during different activities accurately. However, GPS battery life was ~20 h at this resolution, which is less than the average trip duration in some years (Hamer *et al.* 2007; Wakefield *et al.* 2015). Standard-resolution data were not used to examine activity-specific heights but covered much longer time periods, allowing us to track individual birds for up to 7 days and examine broad-scale spatial variation in flight heights during trips. Birds were recaptured and loggers retrieved after ≤10 days (total handling time ~15 min on each capture). After release, birds returned immediately to their nest and resumed normal behaviour. The maximum weight of loggers deployed on birds (48 g) was <2% of body mass (3 kg). Trip durations of instrumented birds were very similar to those of non-instrumented birds observed via a remote radio link using a Mobotix© (Uxbridge, UK) surveillance camera installed in the same area of the colony (Cleasby *et al.* 2015b), and previous studies (e.g. Hamer *et al.* 2000) found that similar devices had no discernible effects on trip durations or body mass.

BAROMETRIC ESTIMATION OF HEIGHT

We used the barometric formula (Berberan-Santos, Bodunov & Pogliani 1997; Wallace & Hobbs 2006) to estimate height z (m) above sea level:

$$z = -\frac{kT}{mg}\ln\left(\frac{P}{P_0}\right), \qquad \text{eqn 1}$$

where P_0 and P are the atmospheric pressures (Pascals) at sea level and at height z (m), respectively; k is the universal gas constant for air (8·31432 N m mol^{-1} K^{-1}); m is the molar mass of air (0·0289644 kg mol^{-1}); g is the acceleration due to gravity (m s^{-1}); and T is the temperature of the atmospheric layer between z_0 and z. Validation of this method using loggers placed at different known heights (see Appendix S1 in Supporting Information for details) indicated that the mean absolute error of height estimated by recording pressure was 0·88 m (range 0·32–1·92 m). Precision of height estimates (σ_z) decreased linearly as the interval, Δt, between observations of P_0 and P increased (Fig. S1). At Δt = 11·5 min (the median value used in our study), σ_z was 1 m.

For both the high-resolution and standard-resolution data sets, we used GPS data to categorize the behaviour of birds based on their location and speed of travel as (i) at the colony, (ii) in flight or (iii) on the water (Wakefield *et al.* 2013). Dives were identified by a rapid increase in pressure above ambient. To calculate birds' heights during periods of flight, pressure data P were smoothed using a running median calculated using a moving window of 11 observations (i.e. over a period of 11 s), centred on each successive location. Prior to smoothing, estimates of pressure ≤5 s before and ≤3 s after dives were removed, as there was typically high variation in pressure within these periods due to acceleration and turbulence. Similarly, following smoothing, we discounted short periods of flight (<3 min) and both the initial and final 60 s of each flight period. For the remaining cases, P_0 was defined where we had high-resolution GPS data as the value of P measured 5 s after the current flight period began. We assumed that whilst at this point the bird would still be flying at low altitude, initial Bernoulli effects associated with increasing airspeed during take-off (Chanson 2009) would have subsided. However, this degree of behavioural discrimination was not possible using the standard-resolution GPS data, so we defined P_0 in these cases as the value of P 10 s before the end of the previous period spent on the water. At the onset of a foraging trip, during the initial bout of flight, P_0 was defined as the pressure at the colony reduced to sea level using eqn. 1 (the height of the colony was known). We also assessed the accuracy of a simpler calibration method in which this value of P_0 was used to estimate height throughout foraging trips.

DATA ANALYSIS

Density of birds at sea

The utilization distribution (UD) of breeding birds from Bass Rock was estimated by calculating the kernel density of bird locations (data sets pooled across all 3 years). Kernel density was estimated on a 1-km^2 grid using the R package adehabitatHR (Calenge 2006). Following Wakefield *et al.* (2013), we then estimated the density of birds within each grid-square (d) as:

$$d = \hat{u}_{i,x}N, \qquad \text{eqn 2}$$

where $\hat{u}_{i,x}$ is the empirical probability density of use of cell x by bird i and N is the size of the colony (75 000 breeding pairs in 2014; Murray, Harris & Wanless 2015). The number of breeding

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pairs was used as a measure of N, rather than the number of breeding individuals, because one member of a pair usually remains at the colony attending the chick (Lewis *et al.* 2004). This approach generated an estimate of the number of breeding gannets expected in each grid cell during the peak chick-rearing period (June–August) each year.

Discrimination of foraging and commuting behaviour

Gannets primarily forage by plunge-diving during daylight (Hamer *et al.* 2009). Following Wakefield *et al.* (2013), we assumed therefore that daytime GPS locations characterized by slow flight speeds and high turning rates indicated foraging (excluding periods when birds were on the water; details and validation of discrimination criteria in Appendix S3). We used the 50% and 95% UDs of foraging locations to represent the core and wider foraging areas, respectively.

Variation in flight heights

We used high-resolution tracking data to examine flight heights during periods of commuting and foraging, and standard-resolution data (i.e. locations at 2-min intervals) to examine spatial variation in flight height (high-resolution data were more precise but included incomplete trips, which could have biased any spatial analysis of heights because birds seldom forage on the return legs of trips; Hamer *et al.* 2009). Flight height was modelled as a log-transformed response variable using a generalized additive mixed model (GAMM) with Gaussian errors in the R package mgcv (Wood 2006). There were some estimated heights ≤0 m (2·7% of values), and these were included in the analysis by adding the minimum estimated height (absolute value) to all cases (to permit log-transformation); data are then presented following back-transformation including subtraction of minimum estimated height in each case. The model included an isotropic spatial smooth of longitude and latitude, permitting spatial predictions of flight height (Wood 2003). Trip identity, nested within bird identity, was included as a random effect. To assess the decay in precision of pressure calibrations over long periods of flight, we also modelled the residual variance in flight height as a function of time elapsed since the most recent calibration of P_0 using the varPower variance function in the nlme R package (Pinheiro & Bates 2000). Flight-height estimates (including negative values) are quoted as medians plus interquartile ranges (IQR).

In addition, we used a binomial GAMM to examine spatial variation in the probability of a bird flying at collision-risk height, that is within the height envelope swept by wind turbine rotors. For this analysis, flight heights between 30 and 160 m above sea level (asl) were scored as 1 (at risk of collision) and all other observations were scored as 0 (not at risk of collision). The risk envelope used here represents a general case rather than a specific turbine design. Fixed and random explanatory covariates were as for the flight-height model described above.

Modelling collision risk

We used a mechanistic collision-risk model (Band 2012; the de facto standard in UK wind farm impact assessments) to assess the potential impact on gannets from Bass Rock of the two closest proposed wind farms within the Firth of Forth area (sites A and B in Fig. 2). Other wind farm developments are also planned,

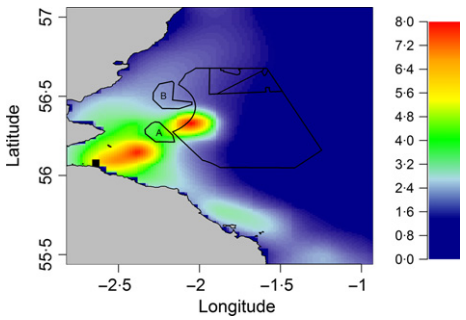


Fig. 2. Estimated density of chick-provisioning northern gannets (birds km^{-2}) in the vicinity of Bass Rock (black square) and the proposed wind farms close to the Firth of Forth (solid lines; sites A and B labelled). Based on all birds equipped with Global Position System-loggers (n = 516 trips by 55 birds).

but our aim was to investigate the importance of accurate flight-height assessments for collision-risk modelling rather than to predict the potential cumulative impact of all proposed wind farms in the region. For comparison, we used both 'basic' and 'extended' versions of the model, treating flight-height data as categorical and continuous, respectively (see Appendix S2 for details).

We ran the models for putative turbine avoidance rates of 98·5% and 99%; a recent comprehensive review of behavioural responses to turbines by different species of seabirds (Cook *et al.* 2014) recommended an avoidance rate of 99% for use with gannets in the basic collision-risk model but considered there were insufficient data to derive a rate for use with the extended model. However, for species with more reliable data on turbine avoidance rates at different spatial scales, the recommended avoidance rate for use in the extended model was consistently 0·5–0·6% lower than that for use in the basic model (Table 7·2 in Cook *et al.* 2014). Hence, taking a precautionary approach, we considered that the lower avoidance rate may be more appropriate for the extended model, although further data are needed to confirm or refute this. To account for the fact that birds do not fly continuously at sea (Hamer *et al.* 2000), we multiplied collision-risk estimates in each case by the proportion of occasions when birds were classified as in flight at each site (average of 82%).

To examine how different methods of recording flight height affected collision-risk estimates, we also ran collision-risk models using published gannet flight heights estimated subjectively by human observers (Cook *et al.* 2012; Johnston *et al.* 2014) in conjunction with our spatial distribution data. Cook *et al.* (2012) modelled gannet flight heights, allowing estimation of the proportion of flights within different collision-risk envelopes, based on ship-based, land-based and radar-based surveys, whilst Johnston *et al.* (2014) presented a continuous gannet flight-height distribution estimated using observer data from 32 wind farm sites. In summary, we compare collision risks estimated: using altimeter data collected during this study in (i) the basic and (ii) the extended models; (iii) using data from Cook *et al.* (2012) in the basic model; and (iv) using data from Johnston *et al.* (2014) in the extended model.

Results

FORAGING TRACKS AND DENSITIES OF BIRDS AT SEA

The foraging ranges of birds extended up to 536·5 km from the colony (mean ± SD = 180·9 km ± 106·0,

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n = 516 trips by 55 individuals) and encompassed proposed wind farm sites close to the Firth of Forth and elsewhere in the North Sea (Fig. 1). Gannet densities during 2010–2012 were high not only close to the Bass Rock but also within proposed wind farm sites close to the Firth of Forth (Fig. 2). Moreover, the foraging UD indicated that the core area of foraging activity (50% foraging UD) overlapped extensively with proposed wind farm sites close to the colony (Fig. 3).

FLIGHT HEIGHTS

High-resolution data

Data collected using GPS and pressure loggers allowed us to reconstruct the three-dimensional movements of birds (Fig. 4). Estimates of height obtained using pressure loggers were significantly positively correlated with GPS-estimated heights (r = 0·58, P < 0·01), with a median precision of ±1 m (see Materials and methods section). We are therefore confident that they accounted appropriately for spatiotemporal variation in atmospheric pressure at sea level. In contrast, using P_0 values recorded at the colony at the beginning of each trip gave estimates that were about 40% higher on average and poorly correlated with GPS-estimated heights (r = 0·23, n.s.). We therefore deemed this technique unreliable and did not consider it further.

Flight height was bimodal, with significantly lower heights during commuting bouts (median = 11·5 m, n = 738, IQR = 6·1–21·7 m) than during foraging bouts (median = 26·5 m, n = 464, IQR = 9·7–46·1 m; randomization test, P < 0·001; see Appendix S4 for details). Hence, foraging birds tended to be at or around collision-risk height (Fig. 5). Although the second peak was discernible across all locations visited by birds (Fig. 5a), it was more prominent within the locations of proposed wind farms (Fig. 5b), where a risk envelope of 20–120 m

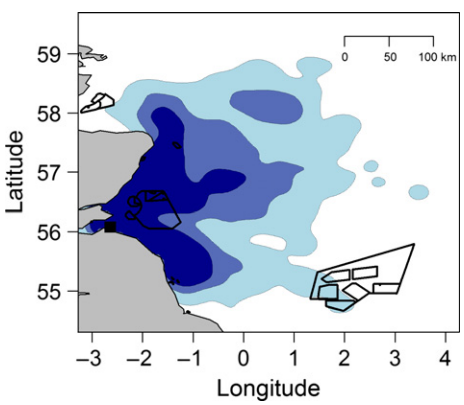


Fig. 3. Foraging utilization distribution (UD) of chick-provisioning northern gannets tracked from Bass Rock (black square) during June–August in 2010–2012, together with locations of proposed wind farm sites. Contours show 50% (darkest), 75% (intermediate) and 95% (palest) UDs.

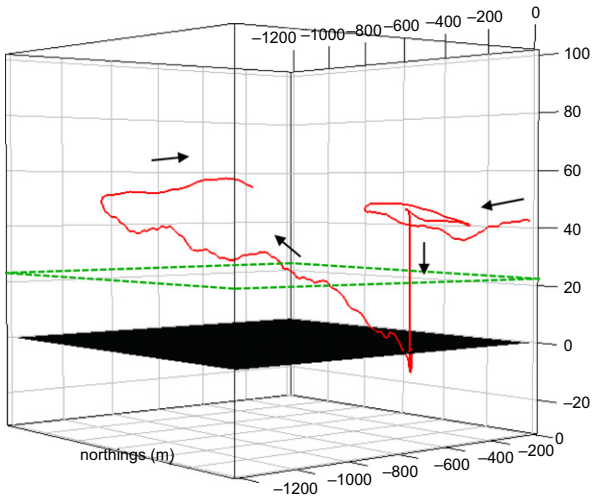


Fig. 4. Five-minute section of a three-dimensional foraging track of a gannet breeding at Bass Rock, recorded using Global Position System (x, y) and pressure difference (z). The bird moves from right to left, circling once, before plunge-diving to 6 m below sea level. On returning to the surface, it takes off and again begins circling. The dotted green line indicates that the bird spent most of this time above the minimum collision-risk height (22 m above sea level).

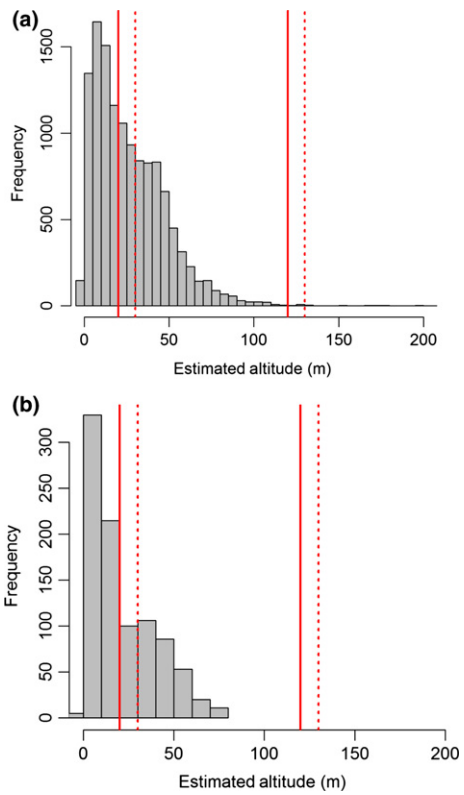


Fig. 5. Distribution of estimated flight heights for (a) all locations visited by birds (n = 16 birds, 12989 observations) and (b) area encompassed by proposed wind farms close to the Firth of Forth (n = 11 birds, 962 observations; five birds equipped with altitude loggers never visited this area). Vertical lines denote two potential collision-risk envelopes (solid lines, 20–150 m above sea level; dashed lines, 30–160 m above sea level).

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asl contained ~10% more observations than one of 30–130 m asl. There was no significant difference between flight heights determined using high-resolution GPS data (median = 13·3 m, n = 1202 height estimates, IQR = 6·9–34·6 m) and those determined using the much larger standard-resolution data set (median = 22·0 m, n = 12621 height estimates, IQR = 10·1–40·0 m; randomization test, P = 0·5; see Appendix S5).

Spatial analysis

There was significant spatial variation in flight heights of birds (Table 1) and in the probability of flying at collision-risk height (Table 2). Areas of high average height were all >100 km from the colony, with lower average heights occurring mainly, though not entirely, inshore and closer to the colony (Fig. 6). There was also significant variation among individuals in flight height (Table 1) and in the probability of flying at collision-risk height (Table 2). Although the precision of our height estimates decreased linearly as the interval between observations of P_0 and P increased (Fig. S1), the time elapsed since measuring P_0 had no effect on the variance of flight-height estimates and including this term in the model of flight height did not improve its performance (Table 1).

COLLISION-RISK ESTIMATES

Mortality predicted using combined altimeter and GPS data was 11·6 times that obtained using the overall distribution of flight heights estimated by observers at sea combined with GPS data (cf. extended models, Table 3) and 5·9 times that obtained by assuming that 5% of birds fly at collision-risk height, based on observer and radar data (cf. basic models, Table 3). Using combined altimeter and GPS data, the basic model with 99% avoidance, which we consider the most reliable estimate, suggested that ~300 breeding adults could be killed per month during the chick-rearing period each year as a result of collisions with turbines at the two planned wind farm sites included in the model (Table 3).

Table 2. Generalized additive model of the probability of gannets flying within a collision-risk envelope of 30–160 m)

Variable	Estimate	Lower 95% CI	Upper 95% CI	*P*-value	ΔAICc
Fixed effects					
Intercept	−0·78	−1·13	−0·42	<0·001	NA
Isotropic smoothers					
s (Lon, Lat)	d.f. = 26·29			<0·001	+948
Random effects					
Trip ID	0·72	0·55	0·94		+98
Bird ID	0·53	0·29	0·96		+592

Discussion

DATA COLLECTION

We found that standard-resolution and high-resolution GPS data (recorded at intervals of 2 min and 1 Hz, respectively), combined with pressure data collected at 1 Hz, resulted in similar gannet flight-height estimates. The use of the standard-resolution GPS-tracking may resolve changes in behavioural state less finely but allows birds to be tracked for longer periods. Time since last calibration had little effect on the precision of our flight-height estimates, perhaps because atmospheric conditions were relatively quiescent during our study. However, it would be desirable to establish how precision decays with this parameter in more dynamic weather conditions, for which further study would be required. Our data show that calibration using pressure recorded in the colony at the beginning of the trip resulted in biased and unreliable estimates of height, so we do not recommend this method, at least in temperate regions or for species that range widely.

The data presented here were all obtained in relatively calm periods, during mid-summer, in years of near-average prey availability (inferred from relative foraging distances; median ± IQR = 175 ± 70 km in 2010, 109 ± 110 km in 2011, 160 ± 145 km in 2012, compared

Table 1. Generalized additive model of gannet flight height (n = 12989 locations from 54 foraging trips by 16 birds)

Variable	Estimate	Lower 95% CI	Upper 95% CI	*P*-value	ΔAICc
Fixed effects					
Intercept	3·02	2·80	3·24	<0·001	NA
Isotropic smooths					
s (Lon, Lat)	d.f. = 21·98			<0·001	+508
Random effects					
Trip ID	0·36	0·23	0·57		+640
Bird ID	0·38	0·30	0·48		+1360
Variance functions					
Time elapsed since P_0 measured	−0·001	−0·004	0·004		−136

ΔAICc is the change in the Akaike Information Criterion (corrected for finite sample sizes) associated with deleting a term from the best fitting model; +ΔAICc indicates that the model fit is worse after deleting a term; ΔAICc <2 indicates that removing the term has little effect on the model. d.f., estimated degrees of freedom for the isotropic smooth, reflecting sinuosity.

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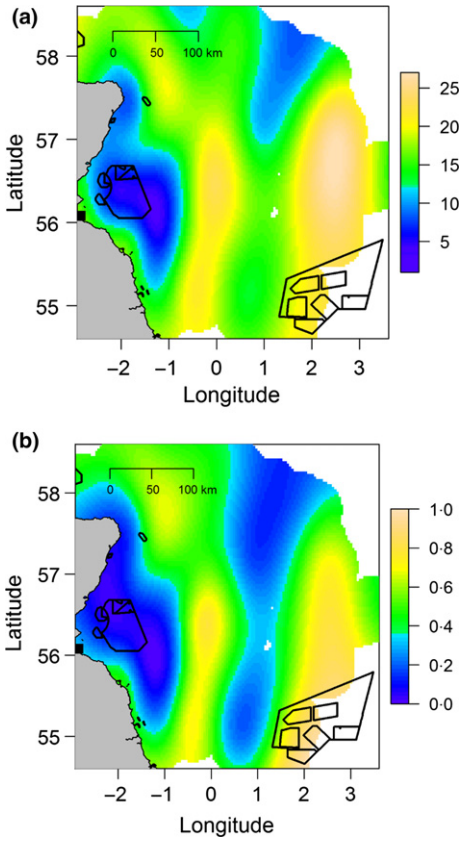


Fig. 6. Variation in (a) mean flight height (m) and (b) the probability of flying within a collision-risk envelope of 30–160 m, across the sampled foraging distribution of gannets breeding at Bass Rock (black square). Locations of proposed wind farms within the foraging range are also shown.

Table 3. Predicted potential collisions per month with wind turbines at two recently consented offshore wind farms, for gannets rearing chicks at Bass Rock

	Flight-height data			Avoidance rate	
Model	Type	Recording method	Source	98·5%	99%
Basic	Categorical*	Ship-/shore-based observers and radar	Cook *et al.* (2012)	80	**53**
	Continuous	Pressure loggers	This study	467	**311**
Extended	Continuous	Ship-based observers	Johnston *et al.* (2014)	**26**	16
	Continuous	Pressure loggers	This study	**304**	203

Figures in bold indicate rates considered most appropriate based on data in Cook *et al.* (2014).
*Taking the proportion of flights within the modelled collision-risk envelope (29–159 m asl) to be 5%; estimated from height distribution of survey data (figure 3·6 in Cook *et al.* 2012).

to long-term average of 155 km, range of annual medians = 105 ± 133–238 ± 193 km; Hamer *et al.* 2007; Wakefield *et al.* 2015). The foraging ranges of gannets vary with population size and per capita prey availability (Hamer *et al.* 2007; Wakefield *et al.* 2013), and their foraging behaviour also varies in relation to weather conditions and feeding opportunities including potential prey species (Amélineau *et al.* 2014; Cleasby *et al.* 2015b). Hence, the potential occurrence of gannets in the turbine envelope of different proposed wind farm sites will also vary with these factors resulting in both spatial and temporal variability in collision risk. Moreover, foraging ranges of breeding birds are frequently longer during incubation than chick-rearing, and the movement patterns of juvenile and immature birds, non-breeding adults and failed breeders are only poorly understood but could also encompass much larger foraging areas than those of breeding birds (Votier *et al.* 2011). Foraging ranges and behaviour also differ between males and females (Cleasby *et al.* 2015b), potentially resulting in an asymmetry in male–female collision risks. In addition, we found that flight heights varied significantly among individuals, in keeping with consistent individual variation in foraging locations, movements, diets and diving behaviour (Patrick *et al.* 2015; Wakefield *et al.* 2015), producing likely marked differences among individuals in collision risk. Further data are needed on all of these factors in order to make a full assessment of the collision risks posed to gannets and other potentially vulnerable species by offshore wind farm developments.

FORAGING AREAS AND FLIGHT HEIGHTS

Using high-resolution (1 Hz) data, we estimated that gannets flew at a median height of 12 m whilst commuting and 27 m during foraging bouts, which corresponds reasonably well with a previous estimate of 37 m whilst foraging (Garthe *et al.* 2014). In contrast, flight heights estimated from ship-based surveys are much lower, with fewer than 10% of flights exceeding 20 m (Cook *et al.* 2012). This discrepancy may partly reflect a high proportion of survey data from sites in the southern North Sea where gannets are mainly seen during migration, when they may spend little time foraging. Radar-based estimates were also low (mean height *c.* 10 m; Parnell *et al.* 2005), probably because they were site-specific with a maximum range within *c.* 6 km of land, meaning that these estimates are likely to represent a high proportion of birds commuting along the coast.

In keeping with spatial density estimates from ship-based surveys, areas with the highest density of gannets at sea were close to the colony (Camphuysen *et al.* 2012) and overlapped proposed wind farm sites close to the Firth of Forth. Many of these birds were apparently commuting to or from foraging areas further from the colony, at heights typically below 15 m, resulting in a relatively low probability of flying at collision-risk height. However,

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gannets also forage during the outward portions of trips (Hamer *et al.* 2009), increasing their potential collision risk. Combined with the high density of birds close to the colony, this resulted in a large number of potential collisions within proposed wind farm sites despite a low average flight height. Further from the colony, average heights were greater as a result of relatively little commuting flight in the distal sections of trips, probably combined with spatial variation in foraging behaviour; birds rely on momentum to attain depth during V-shaped dives and these were significantly deeper in stratified water offshore than in mixed inshore waters close to the colony (Cleasby *et al.* 2015b), suggesting that birds were diving from greater heights offshore.

POTENTIAL POPULATION-LEVEL EFFECTS

Using our data on flight heights, the basic Band (2012) collision-risk model with 99% avoidance and the extended model with 98·5% avoidance yielded very similar estimates for the two sites in our analysis, predicting that during chick-rearing, 311 and 304 breeding adults, respectively, could be killed each month. There is great uncertainty over actual avoidance rates (Cook *et al.* 2014), but taking a figure of ~300 collisions per month and assuming similar foraging behaviour during incubation and chick-rearing, the cumulative predicted mortality during the breeding season (mid-April to mid-September) each year would be ~1500 adults.

To put these data into perspective, a population model for gannets at Bass Rock, based on a population of 48 000 breeding pairs in 2004, suggested that additional mortality of 2000 birds per year, estimated to comprise 1400 adults and 600 immature birds, would be sufficient to cause a sustained decrease in breeding population size (WWT Consulting 2012). This threshold may have been underestimated, since the breeding population at Bass Rock has apparently increased by an average of 2700 pairs per year since then (Murray, Harris & Wanless 2015). Nonetheless, our estimate of predicted adult mortality due to collision with turbines is sufficiently high to cause concern. Moreover, it may be conservative: adults are present at the colony for 9 months each year (Nelson 2002), they may be killed at other proposed wind farms not considered in our study, and we assume a minimum blade clearance of 30 m, whereas this may be to be up to 8 m lower (Marine Scotland 2014). Hence, there is an urgent need for further data, both for gannets and for other high-priority species such as large gulls, to refine collision-risk estimates and mortality thresholds for long-term population viability, and for strategic monitoring at key sites to determine whether predicted collision mortality is realized and has significant effects on population trajectories. We also strongly recommend that, at sites with high potential collision risk, the minimum permitted clearance of turbine blades should be raised from 22 to 30 m above sea level.

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Data accessibility

GPS logger and barometric logger data are available free of charge via Birdlife International's Seabird Tracking Database (http://seabirdtracking.org/mapper/contributor.php?contributor_id=204) and the Dryad Digital Repository doi: 10.5061/dryad.1ds1q (Cleasby *et al.* 2015a).

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Appendix S1. Validation of barometric altitude estimates.

Appendix S2. Discrimination of foraging and commuting behaviour.

Appendix S3. Statistical comparison of foraging and commuting flight heights.

Appendix S4. Statistical comparison of flight heights calculated using high-resolution and standard-resolution data.

Appendix S5. Modelling collision risk.

Table S1. Data used in mechanistic collision-risk models.

Fig. S1. Precision of height estimates based on pressure recorded at the Bass Rock vs. interval between pressure observations.

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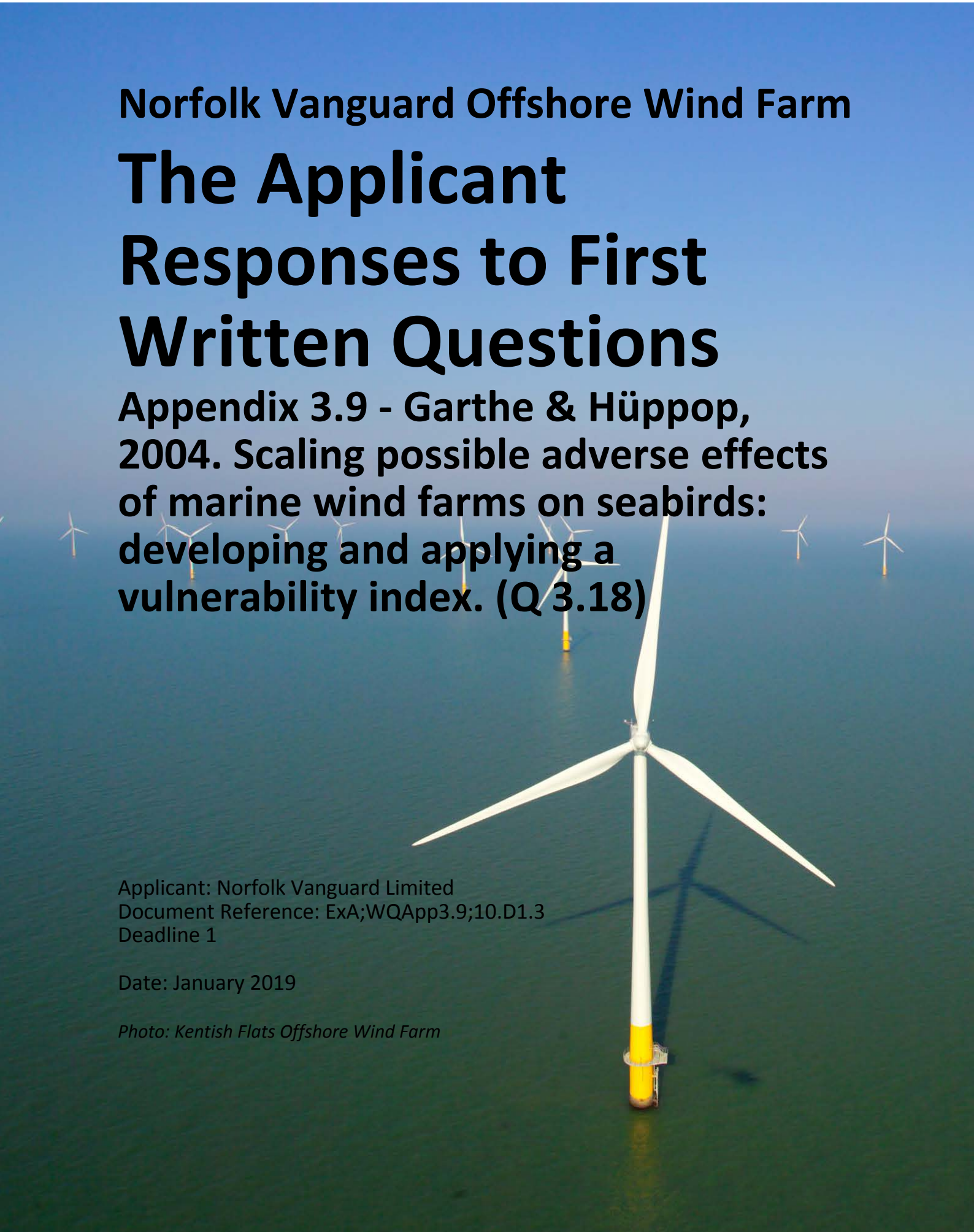
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The Applicant Responses to First Written Questions

Appendix 3.9 - Garthe & Hüppop, 2004. Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index. (Q 3.18)



Applicant: Norfolk Vanguard Limited
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Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index

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Summary

1. Marine wind farms have attracted substantial public interest. The construction of wind facilities offshore may become Europe's most extensive technical development in marine habitats. Due to political pressure to complete construction soon, assessments of possible wind farm locations, for example in the German sectors of the North Sea and Baltic Sea, have to be based on existing knowledge.

2. In this study, we developed a wind farm sensitivity index (WSI) for seabirds. We applied this index to the Exclusive Economic Zone and the national waters of Germany in the North Sea. We chose nine factors, derived from species' attributes, to be included in the WSI: flight manoeuvrability; flight altitude; percentage of time flying; nocturnal flight activity; sensitivity towards disturbance by ship and helicopter traffic; flexibility in habitat use; biogeographical population size; adult survival rate; and European threat and conservation status. Each factor was scored on a 5-point scale from 1 (low vulnerability of seabirds) to 5 (high vulnerability of seabirds). Five of these factors could be dealt with by real data but four could only be assessed by subjective considerations based on at-sea experience; in the latter cases, suggestions of the first author were independently modulated by experts.

3. Species differed greatly in their sensitivity index (SSI). Black-throated diver *Gavia arctica* and red-throated diver *Gavia stellata* ranked highest (= most sensitive), followed by velvet scoter *Melanitta fusca*, sandwich tern *Sterna sandvicensis* and great cormorant *Phalacrocorax carbo*. The lowest values were recorded for black-legged kittiwake *Rissa tridactyla*, black-headed gull *Larus ridibundus* and northern fulmar *Fulmarus glacialis*.

4. A WSI score for areas of the North Sea and Baltic Sea was calculated from the species-specific sensitivity index values. Coastal waters in the south-eastern North Sea had values indicating greater vulnerability than waters further offshore throughout the whole year.

5. Derived from the frequency distribution of the WSI, we suggest a 'level of concern' and a 'level of major concern' that are visualized spatially and could act as a basis for the selection of marine wind farm locations.

6. *Synthesis and applications.* The wind farm sensitivity index might be useful in strategic environmental impact assessments (EIA). Results of small-scale EIA from wind installations should be considered within a more global perspective, provided, for example, by large mapping projects and detailed behavioural studies. This is difficult in normal EIA, particularly in highly dynamic coastal/marine habitats, and the results of this study fill an important gap by providing information on the potential sensitivity of seabirds and the importance of locations of wind installations.

Key-words: birds, conservation, management, sensitivity analysis, wind energy

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Introduction

The first wind farms at sea were established in the early 1990s, off Denmark and Sweden (Larsson 1994). There are now at least nine operational marine wind farms in Europe, as well as proposals to build many more around the UK and off Germany and Denmark (Anonymous 2000a,b, 2001, 2002; ICES 2003). Plans for marine wind farms indicate that each may consist of up to 1000 turbines, extending as far offshore as 100 km, and in waters up to 40 m deep (Anonymous 2002). Within north-west Europe, there are in total about 260–270 turbines in existing marine wind farms (as of December 2003, http://www.offshorewindenergy.org/) but many thousands are planned for construction in the next few years (Hüppop, Exo & Garthe 2002; ICES 2002). According to the current development plan for the German parts of the North and Baltic Seas, marine wind farms will require an area of 13 000 km^2 between 2002 and 2030 (BMU 2001). The UK has recently issued licenses for the development of about 40 marine wind farms in its waters. Thus, erection of wind facilities offshore may become Europe's most extensive technical development in marine habitats (Merck & von Nordheim 2000).

With so few wind farms established in the sea to date, there is very limited information on their effects on the marine environment (Merck & von Nordheim 2000; ICES 2002) and none on marine sites located more than 10–20 km from the coast. As applications for offshore wind farm construction will be decided in many locations before comprehensive, medium- to large-scale, ecological studies on the status of marine wildlife are completed, predicted effects have to be based on limited current knowledge. This should include conclusions from studies of wind farms on land, from the few inshore wind farms and from knowledge of the spatiotemporal patterns of abundance of organisms at sea that might be at risk.

Birds are assumed to be among the taxa affected most heavily by wind farms. Studies on land, and the first results from marine sites, suggest that both birds on migration and those resting or foraging locally may be affected (Barrios & Rodríguez 2004). At sea, this therefore includes both migrating birds, from the smallest songbirds to large birds such as cranes and birds of prey, and seabirds during their local movements (Anonymous 2000b; Garthe 2000; Exo, Hüppop & Garthe 2003; Hüppop, Exo & Garthe 2002). From extensive studies of seabirds at sea over the past 20 years, the distribution and abundance of seabirds in the North Sea is well-known over large and medium scales. Available data can therefore advise the site selection of wind farm locations. However, different habits and activities of birds at sea have to be taken into account. For example, species flying frequently at altitudes of 50–200 m a.s.l. are much more vulnerable to wind turbines than species that swim most of the time. However, there are many effects to be considered in addition to direct collisions.

In order to assess the possible impacts on seabirds of a range of factors, several indices have been applied in recent years, for example with regard to oil pollution (King & Sanger 1979; Williams *et al*. 1994) and the sandeel fishery (Furness & Tasker 2000). One of the indices of vulnerability to oil pollution has been applied to the whole North Sea, separating areas of high and low vulnerability for seabirds over the year (Carter *et al*. 1993). While the lack of long-term data, the inability to infer causality from monitoring studies and the limited spatial and temporal scales of experimental studies make such indices difficult to derive with confidence, more are required (Forde 2002). Debate on the effects of human activities on wildlife necessitates risk and impact assessments (Stillman *et al*. 2001) even where the database might be poor (Tuck *et al*. 2001).

In this study, we developed a wind farm sensitivity index (WSI) for seabirds. In terms of rationale and methodology, this index follows those of Williams *et al*. (1994) and Furness & Tasker (2000). We applied this index to the Exclusive Economic Zone and national waters of Germany in the North Sea. This area was selected for three reasons. First, there has not been any governmental advice prior to the beginning of the planning process (as was the case in Denmark, for example; Anonymous 1996) so site selection by the applicants might not have taken into account all of the important environmental issues. Secondly, there is an urgent need to evaluate risks in this area, in which 24 project applications comprising more than 11 000 single turbines were in place as of January 2003 (in the Exclusive Economic Zone). Thirdly, due to the complex geomorphology and hydrography under the existing intensive human usage (e.g. shipping lanes), a small- and medium-scale environmental assessment is urgently needed as a solid basis for future licensing.

Methods

VULNERABILITY FACTORS

We chose nine different factors, derived from species' attributes, to be included in the WSI, all taking into account the risks of seabirds colliding with wind turbines and/or being disturbed by wind farms. Each factor was scored on a 5-point scale from 1 (low vulnerability to seabirds) to 5 (high vulnerability to seabirds). Five of these factors could be based on real data, four could only be assessed by subjective considerations based on at-sea experience. In the latter cases, our suggestions were sent for independent evaluation by 10 experts per factor. The experts were chosen according to their experience (e.g. in ship-based and aerial seabird surveys) from a total of eight national and five international experts (listed in the Acknowledgements). Following Furness & Tasker (2000), we made changes if two or more experts suggested alterations to the original categorization in the same direction. The nine factors included are outlined below.

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(a) Flight manoeuvrability

This factor took into account flight properties with regard to the potential to avoid collision with wind farms at sea. It was assessed subjectively, based on extensive field experience, and was modulated by experts as described above. Species were ranked from a very high flight manoeuvrability (score 1) to low flight manoeuvrability (score 5). A fast-flying, relatively heavy species such as the common guillemot *Uria aalge* (Pontoppidan) is thus considered much more vulnerable compared with a very agile species such as the Arctic tern *Sterna paradisaea* Pontoppidan, which is assumed to be able to escape wind turbines much better.

(b) Flight altitude

This factor was based on flight altitude assessments made during regular seabirds at sea surveys (see Distributional data). Flight altitudes were estimated using binoculars, distance meters and comparative height measures on the ships according to the following height classes: 1, 0–5 m; 2, 5–10 m; 3, 10–20 m; 4, 20–50 m; 5, 50–100 m; 6, > 100 m. A further separation of high flight altitudes was not useful. Flight altitude data were converted to a 5-point scale by using two different percentiles of the flight altitude data distributions, the median (= 50 percentile) and 90 percentile. The 90 percentile was chosen in addition to the median to take into account the few birds that flew high (i.e. 90% of the birds flew in the same or lower height classes, 10% of the birds flew in the same or upper height classes). The scores were classified as follows: 1, height class 1 for the median; 2, height class 2 for the median; 3–5, height class 3 for the median but the 90 percentile differed, score 3 had height classes < 5 for the 90 percentile, score 4 height class 5 for the 90 percentile and score 5 height class 6 for the 90 percentile.

(c) Percentage of time flying

The percentage of time flying was obtained from seabirds at sea counts, with numbers of swimming birds corrected for individuals overlooked at larger distances (see below). Species were scored 1 if 0–20% of the individuals in the transect were flying, 2 if 21–40% in the transect were flying, 3 if 41–60%, 4 if 61–80% and 5 if 81–100% of the individuals in the transect were flying.

(d) Nocturnal flight activity

Nocturnal flight activity could not be quantified by real data and was thus classified subjectively from 1 (hardly any flight activity at night) to 5 (much flight activity at night). Information for this classification was taken from comprehensive handbooks such as Glutz von Blotzheim & Bauer (1982) and Cramp & Simmons (1983). Field experience as well as personal observations were also used (Garthe & Hüppop 1996). Our classification was subsequently modulated by experts as indicated above.

(e) Disturbance by ship and helicopter traffic

Species react differently to the ship and helicopter traffic that occurs during the construction and maintenance of wind farms. Such behaviour might also give an indication of the general behaviour of birds towards disturbances. Due to the paucity of data, this factor was scored subjectively from 1 (hardly any escape/avoidance behaviour and/or none/very low fleeing distance) to 5 (strong escape/avoidance behaviour and/or large fleeing distance) and classifications were modulated by experts. Our scores resulted from extensive surveys at sea from boats, where the reactions of birds to the approaching platform were experienced constantly. Also, species' reactions to counts from aerial surveys (low-flying aeroplanes) as well as to over-flying aeroplanes and helicopters in coastal areas were used.

(f) Flexibility in habitat use

Habitats at sea are often defined by hydrographic characteristics. Because these hydrographic characteristics, e.g. water masses and fronts, depend on wind direction and speed as well as tidal stage, they often vary in location and may shift over many tens of kilometres. This factor scored the flexibility in habitat use of the different species. It could only partially be based on real data, such as published in Garthe (1997) and Skov & Prins (2001). Thus, in addition, unpublished data on seabird–habitat relationships were analysed. We scored this factor from 1 (very flexible in habitat use) to 5 (reliant on specific habitat characteristics) based on the information sources listed above. Species scored low were those occupying large sea areas with no specific habitat preference (e.g. lesser black-backed gull *Larus fuscus* Linnaeus), while species that scored high were those relying on specific habitat features (e.g. sea ducks occurring over bivalve banks on shallow grounds). Our classifications were again modulated by experts.

(g) Biogeographical population size

This factor was scored according to the respective biogeographical population size of each species. Population sizes were taken, if available, from either Rose & Scott (1997) or by collating area-specific data species by species from Lloyd, Tasker & Partridge (1991). Score 1 was given for population sizes exceeding 3 million individuals; score 2 for > 1 million up to 3 million individuals; score 3 for > 500 000 up to 1 million individuals; score 4 for > 100 000 up to 500 000 individuals; and score 5 for less than 100 000 individuals.

(h) Adult survival rate

As additional mortality due to collisions affects species with high annual survival rates more than species with low survival rates, we included this factor. A score of 1 was given if the annual survival rate ≤ 0·75; 2, > 0·75–0·80;

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Table 1. Annual adult survival rates of the bird species involved in the index

Species name (English)	Species name (scientific)	Annual adult survival	Source
Red-throated diver	*Gavia stellata*	0·84	Hemmingsson & Eriksson (2002)
Black-throated diver	*Gavia arctica*	0·85	Nilsson (1977), Hemmingss on & Eriksson (2002)
Great crested grebe	*Podiceps cristatus*	0·7	Fuchs (1982)
Red-necked grebe	*Podiceps grisegena*	0·7	Estimate
Northern fulmar	*Fulmarus glacialis*	0·986	del Hoyo, Elliott & Sargatal (1992)
Northern gannet	*Morus bassanus*	0·94	del Hoyo, Elliott & Sargatal (1992)
Great cormorant	*Phalacrocorax carbo*	0·84	Krementz, Sauer & Nichols (1989)
Common eider	*Somateria mollissima*	0·895	Krementz, Barker & Nichols (1997)
Black scoter	*Melanitta nigra*	0·773	Krementz , Barker & Nichols (1997)
Velvet scoter	*Melanitta fusca*	0·77	Estimate
Arctic skua	*Stercorarius parasiticus*	0·84	del Hoyo, Elliott & Sargatal (1996)
Great skua	*Catharacta skua*	0·90	del Hoyo, Elliott & Sargatal (1996)
Little gull	*Larus minutus*	0·80	Estimate
Black-headed gull	*Larus ridibundus*	0·825	Glutz von Blotzheim & Bauer (1982)
Mew gull	*Larus canus*	0·80	Glutz von Blotzheim & Bauer (1982)
Lesser black-backed gull	*Larus fuscus*	0·93	Wanless *et al*. (1996)
Herring gull	*Larus argentatus*	0·93	Glutz von Blotzheim & Bauer (1982)
Great black-backed gull	*Larus marinus*	0·93	Glutz von Blotzheim & Bauer (1982)
Black-legged kittiwake	*Rissa tridactyla*	0·81	del Hoyo, Elliott & Sargatal (1996)
Sandwich tern	*Sterna sandvicensis*	0·88	Estimate
Common tern	*Sterna hirundo*	0·88	del Hoyo, Elliott & Sargatal (1996)
Arctic tern	*Sterna paradiasea*	0·875	del Hoyo, Elliott & Sargatal (1996)
Black tern	*Chlidonias niger*	0·88	Estimate
Common guillemot	*Uria aalge*	0·885	del Hoyo, Elliott & Sargatal (1996)
Razorbill	*Alca torda*	0·905	del Hoyo, Elliott & Sargatal (1996)
Atlantic puffin	*Fratercula arctica*	0·95	del Hoyo, Elliott & Sargatal (1996)

3, > 0·80–0·85; 4, > 0·85–0·90; 5, > 0·90. For survival rates see Table 1. Due to a lack of data, for red-necked grebe *Podiceps grisegena* (Boddaert), velvet scoter *Melanitta fusca* (Linnaeus), little gull *Larus minutus* Pallas, sandwich tern *Sterna sandvicensis* Latham and black tern *Chlidonias niger* (Linnaeus), values from closely related species had to be taken.

(i) European threat and conservation status

This factor reflected both threat and conservation status of the species in Europe as given by Tucker & Heath (1994). Species were scored 1 if the threat status was 'secure' and no species of European concern (SPEC) status given. A score of 2 was given for species with a threat status of 'secure' but a SPEC status of 4 (species whose global populations are concentrated in Europe). Species judged 'localized' for threat status were scored 3, those listed as 'declining' 4 and those judged 'vulnerable' 5.

SENSITIVITY INDEX CALCULATION

We organized the nine vulnerability factors into three groups, comprising (A) flight behaviour (factors a–d), (B) general behaviour (factors e–f) and (C) status (factors g–i). For each group, an average score of the respective factors was calculated. These average scores were subsequently multiplied by each other to give the species-specific sensitivity index (SSI) for each species:

$$\mathrm{SSI} = \frac{(a+b+c+d)}{4} \times \frac{(e+f)}{2} \times \frac{(g+h+i)}{3}$$

DISTRIBUTIONAL DATA

Distribution at sea was assessed by counts from boats following the methods of Tasker *et al*. (1984), Webb & Durinck (1992) and Garthe, Hüppop & Weichler (2002). Transects were always 300 m wide and were set to one or both sides of the vessels. Because some birds were overlooked in the outer areas of the transect, the density of swimming birds was corrected using the values provided by Stone *et al*. (1995). For grebes, being quite rare in North Sea waters and not dealt with by Stone *et al*. (1995), we used a correction factor of 1·3 based on our own, more extensive, data sets from the western Baltic Sea. The density of flying birds was not corrected, assuming that flying birds were recorded more or less completely within the transects (Stone *et al*. 1995; Garthe 1997). Data originated from the European Seabirds at Sea Database version 3.0 (July 2002) and the German Seabirds at Sea Database version 3.06 (April 2003). Databases are described in Stone *et al*. (1995) and Garthe, Hüppop & Weichler (2002).

VULNERABILITY MAPS

Seabird vulnerability to offshore wind farms is presented in maps with grids of 6′ latitude × 10′ longitude each, amounting to a total grid size of *c*. 120 km^2. Only data collected under good detectability conditions (in

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sea states 0–4; Garthe, Hüppop & Weichler 2002) from January 1993 to May 2003 were used for analyses. Data were summarized per season: summer = June–August, autumn = September–November, winter = December–February, spring = March–May. Coverage was not equal across the study area. Thus, data were corrected for different survey effort: for each species, the density per grid cell was obtained by dividing the sum of individuals recorded in the transect by the total transect area covered by cruises. To reduce bias due to only short visits to some grid cells, all grid cells with less than 1 km^2 covered were excluded. For each grid cell with sufficient data, the vulnerability was determined as:

$$WSI = \sum_{species} (\ln (density_{species} + 1) \times SSI_{species})$$

Thus, for each species, the respective SSI value was multiplied with the natural logarithm of its density (+1, to avoid undefined values) and subsequently summed over all species.

We also defined three levels for a final evaluation of the area under investigation. The levels were established with the following considerations. The median divides the whole sample into two equally large parts, i.e. half of the grid cells have a WSI larger than the 'average'. By definition these are the areas with a wind farm vulnerability higher than the average. To end up with a more conservative estimate for areas of 'concern' we decided to use the 60 percentile rather than the 50 percentile (= median). Accordingly we assumed a 'level of major concern' for the fifth of all grid cells with the highest WSI indices.

SENSITIVITY ANALYSIS FOR THE WSI

In order to verify how the WSI might be affected by inaccurate scores for any of the nine factors listed above, a sensitivity analysis was carried out. We chose three species, one with a high SSI, one with a medium SSI and one with a low SSI. Randomly, each score for any of the eight factors was altered. In a first run, the scores were altered either by upgrading or downgrading them by 1 (determined by random and only if applicable, e.g. score 5 could not be increased and thus remained). In a second run, the scores were altered either by upgrading or downgrading them by 2 (again determined randomly and only to the extent possible).

Results

SENSITIVITY INDEX

The species had strongly differing sensitivity index values (Table 2). Black-throated diver *Gavia arctica* (Linnaeus) and red-throated diver *Gavia stellata* (Pontoppidan) ranked highest (i.e. were most sensitive), followed by velvet scoter, sandwich tern and great cormorant *Phalacrocorax carbo* (Linnaeus). The lowest SSI values were

Table 2. Score of the nine vulnerability factors and the resulting species sensitivity index (SSI) values for each of the 26 seabird species. For details see text

Bird species	Flight manoeuvrability	Flight altitude	% flying	Nocturnal flight activity	Disturbance by ship and helicopter traffic	Habitat use flexibility	Biogeographical population size	Adult survival rate	European threat and conservation status	SSI
Black-throated diver	5	2	3	1	4	4	4	3	5	44·0
Red-throated diver	5	2	2	1	4	4	5	3	5	43·3
Velvet scoter	3	1	2	3	5	4	3	2	3	27·0
Sandwich tern	1	3	5	1	2	3	4	4	4	25·0
Great cormorant	4	1	4	1	4	3	4	3	1	23·3
Common eider	4	1	2	3	3	4	2	4	1	20·4
Great crested grebe	4	2	3	2	3	4	4	1	1	19·3
Red-necked grebe	4	2	1	1	3	5	5	1	1	18·7
Great black-backed gull	2	3	2	3	2	2	4	5	2	18·3
Black tern	1	1	4	1	2	3	4	4	4	17·5
Common scoter	3	1	2	3	5	4	2	2	1	16·9
Northern gannet	3	3	3	2	2	1	4	5	3	16·5
Razorbill	4	1	1	1	3	3	2	5	2	15·8
Atlantic puffin	3	1	1	1	2	3	2	5	5	15·0
Common tern	1	2	5	1	2	3	3	4	1	15·0
Lesser black-backed gull	1	4	2	3	2	1	4	5	2	13·8
Arctic tern	1	1	5	1	2	3	3	4	1	13·3
Little gull	1	1	3	2	1	3	5	2	4	12·8
Great skua	1	3	4	1	1	2	5	4	2	12·4
Common guillemot	4	1	1	2	3	3	1	4	1	12·0
Mew gull	1	3	2	3	2	2	2	2	4	12·0
Herring gull	2	4	2	3	2	1	2	5	1	11·0
Arctic skua	1	3	5	1	1	2	4	3	1	10·0
Black-headed gull	1	5	1	2	2	2	1	3	1	7·5
Black-legged kittiwake	1	2	3	3	2	2	1	3	1	7·5
Northern fulmar	3	1	2	4	1	1	1	5	1	5·8

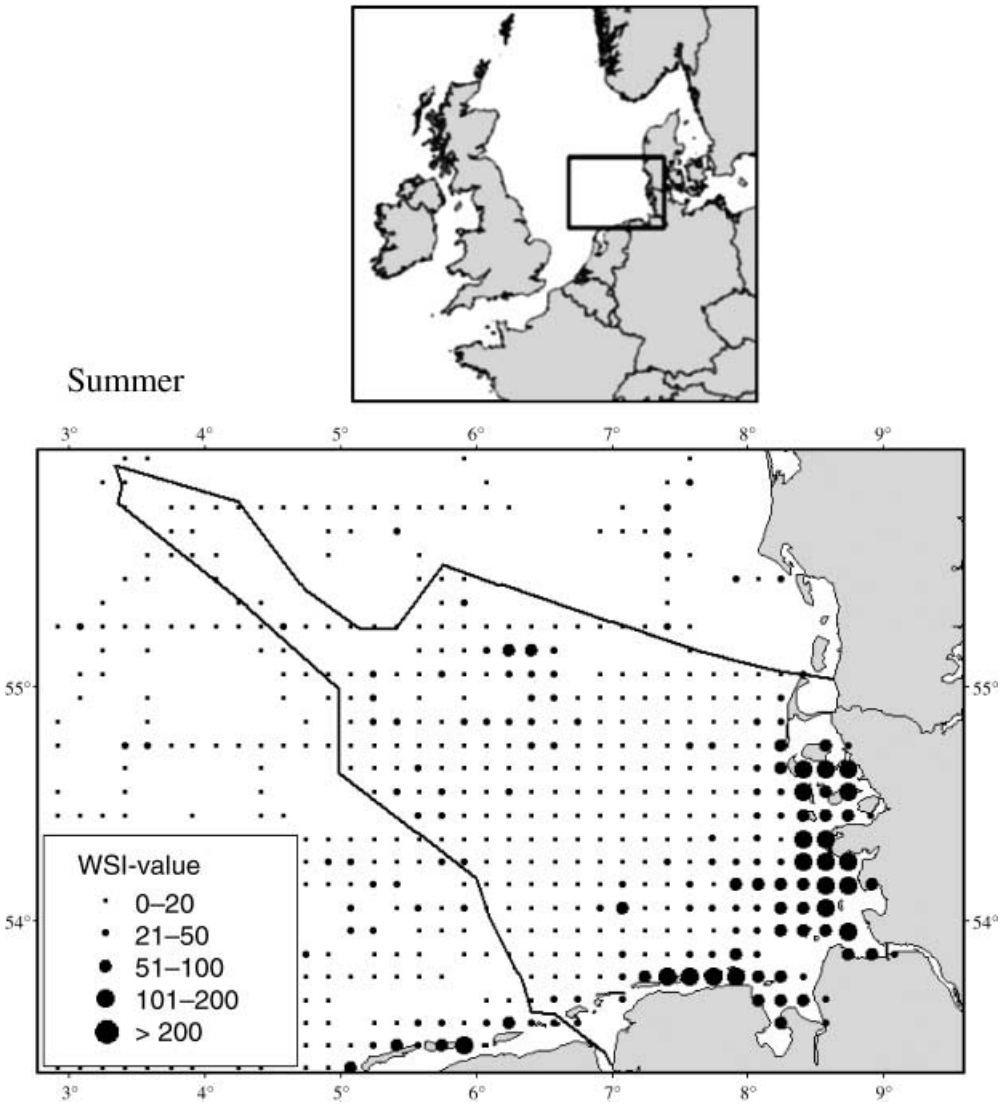


Fig. 1. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in summer (June–August) 1993–2002. For assumptions and calculations see text.

calculated for black-legged kittiwake *Rissa tridactyla* (Linnaeus), black-headed gull *Larus ridibundus* Linnaeus and northern fulmar *Fulmarus glacialis* (Linnaeus).

Sensitivity analyses for the three species selected resulted in moderate deviation when the scores where randomly altered by 1 (mean of 10 runs per species). The SSI for red-throated diver changed from 43·3 to 44·8, the SSI for common eider *Somateria mollissima* (Linnaeus) from 20·4 to 23·0 and that for black-headed gull from 7·5 to 10·0. When the scores were randomly altered by 2, the changes were more pronounced (mean of 10 runs per species). The SSI for red-throated diver was reduced from 43·3 to 24·4, the SSI for common eider increased from 20·4 to 30·3 and that for black-headed gull from 7·5 to 16·4.

AREAS OF VULNERABILITY

Throughout the whole year, WSI values in coastal waters of the south-eastern North Sea were considerably higher than those of waters further offshore. Focusing on the German sector, the coastal zone had consistently moderate to high WSI values in summer (Fig. 1). The area around Helgoland showed some moderate values whereas vulnerabilities further offshore were low. In autumn, WSI values were generally lower than in summer, but a number of coastal sites reached moderate to high vulnerabilities (Fig. 2). Some offshore areas gained importance compared with summer because species' distributions were less confined to breeding sites so that they were more widely distributed. Also, autumn migration certainly created higher densities in areas far away from the coast. In winter, the south-easternmost part of the German Bight was less vulnerable than in summer and autumn (Fig. 3). Nevertheless, it was obvious that WSI values in the coastal zone in winter were usually moderate to high whereas the values far away from the coast were low or very low. In spring, vulnerabilities were again quite high in most areas of the coastal zone but were also moderate to high in areas up to 70–80 km off the northern part of the German coast (Fig. 4).

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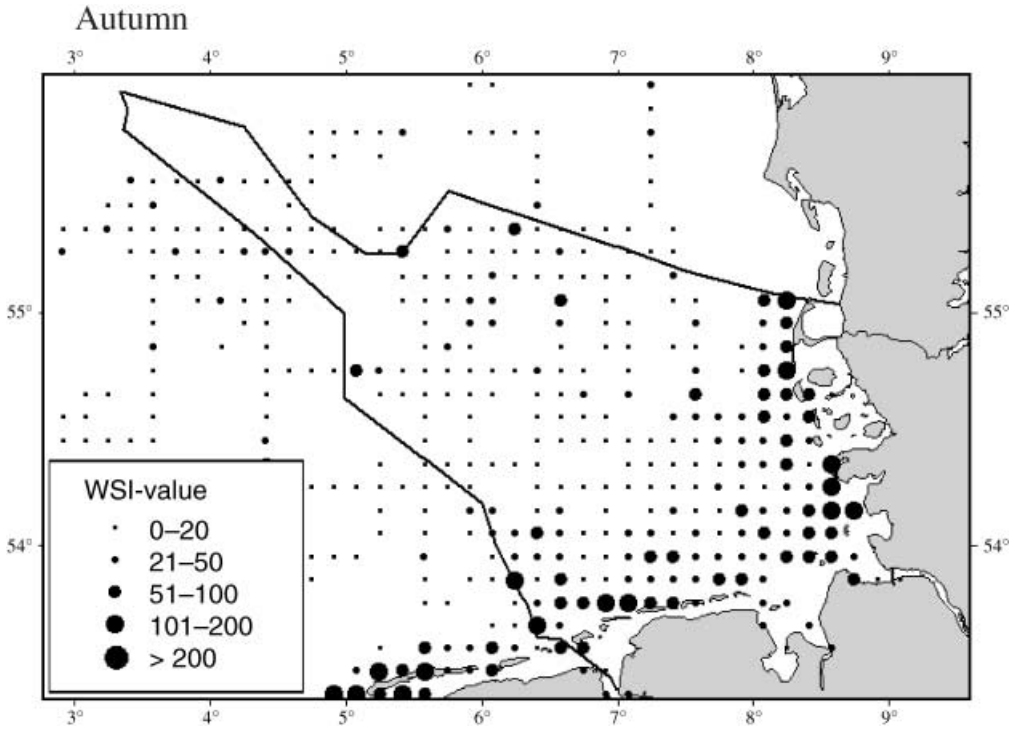


Fig. 2. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in autumn (September–November) 1993–2002. For assumptions and calculations see text.

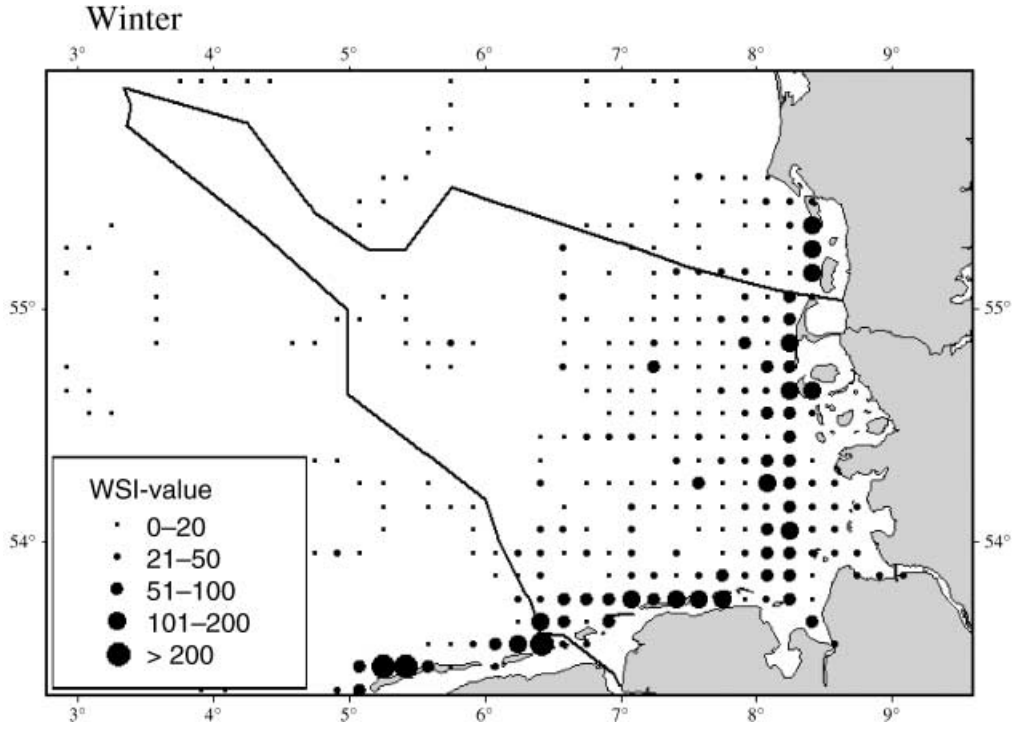


Fig. 3. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in winter (December–February) 1993–2003. For assumptions and calculations see text.

Discussion

SUITABILITY OF THE WSI

Five factors incorporated into the WSI were based on real data but this was not possible for the remaining four factors. We therefore decided to assess these factors subjectively based on at-sea experience. Experts then evaluated our initial scores by reading and considering our values presented on a list, based on their own experiences related to that topic. Such a procedure, called the Delphi technique, has been applied broadly

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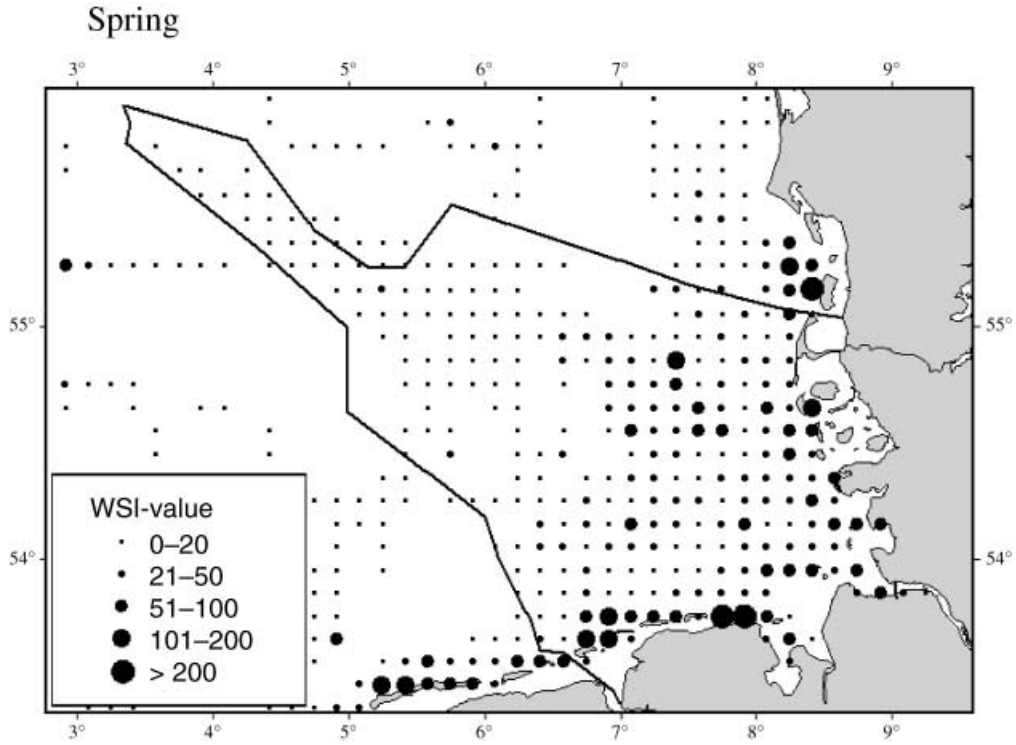


Fig. 4. Spatial distribution of the wind farm sensitivity index (WSI) values (all seabird species combined) in the south-eastern North Sea in spring (March–May) 1993–2003. For assumptions and calculations see text.

before, for example in habitat suitability indices of the US Fish and Wildlife Service (Crance 1987). Recently Cowling *et al*. (2003) compared an expert-based and a systematic algorithm-based approach to identifying priority areas for conservation in the Cape Floristic Region. They concluded that 'rather than emphasize the dichotomy between expert and systematic approaches, conservation planners should devise ways of integrating them'. Although not without difficulties, these data based on expert judgement are currently the best available.

In general, indices depend strongly on the factors selected and the way they are weighed against each other. The WSI proposed in this paper is no exception in this respect. Nevertheless, there are at least four reasons why this index seems to be well-suited to fulfil the urgent need to assess the vulnerability of all seabirds that occur in a large area. First, the final SSI values show substantial differences between the species. Hence, the WSI combines numerical abundance data with evaluations of the sensitivity and importance of the different species. Secondly, not too many alterations were required on the basis of the evaluations made by national and international experts (four changes for flight manoeuvrability, nine changes for nocturnal flight activity, 18 changes for disturbance by ship and helicopter traffic and 15 changes for habitat use flexibility), suggesting that the species-specific scores for each vulnerability factor were well chosen. Interestingly, it has been shown by morphometric measurements and behavioural observations in the literature (Verbeek 1977; Camphuysen 1995) that lesser black-backed gulls have a higher flight manoeuvrability than herring gulls *Larus argentatus* Pontoppidan, providing an example of how accurate our expert judgement system was. Thirdly, sensitivity analyses showed that minor changes in the scores did not affect the SSI much, although major changes may do so. Fourthly, the spatial representation of the WSI values fits well with previous evaluations of the location of important bird areas (Skov *et al*. 1995).

SUGGESTIONS FOR CONSERVATION AND MANAGEMENT

In the German Bight, seabird vulnerability towards wind farms seems to be a function of distance to the coast. The highest values were found relatively close to the coast and lowest values (very) far from the shore. However, there are differences between the two coasts. North of the East Frisian Islands (the southern part of the German sector), the values decrease at much shorter distances from the coast than they do west of the North Frisian Islands (in the eastern part of the German sector). These patterns originate from the density distribution of all species, with a bias towards those species that are ranked high in the SSI and with less emphasis towards species having a low SSI. However, in no case did high densities of a single species create very high WSI values. Thus, the vulnerability maps are more a summary of all species contributing to the final WSI value to a different degree. Species' attributes do not exhibit much variation over the study area so that they contribute primarily via the input factors for the SSI. Thus, spatial variation of the WSI values is basically a reflection of summarized species' density distributions.

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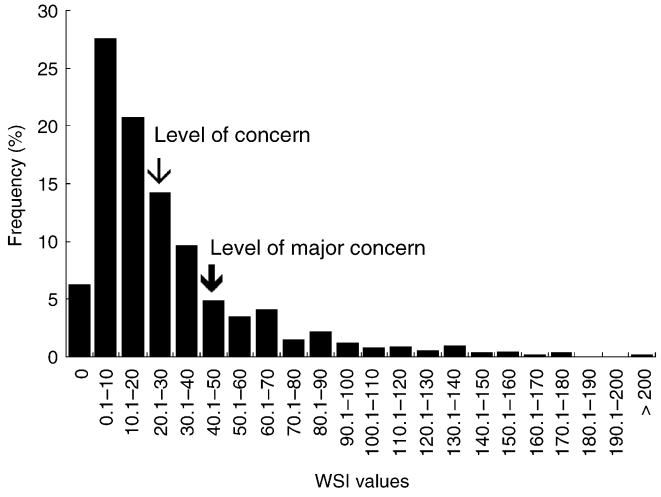


Fig. 5. Average frequency distribution (in percentage) of the WSI values in the German sector of the North Sea. The values were obtained by calculating means per size class over the fours seasons presented in Figs 1–4.

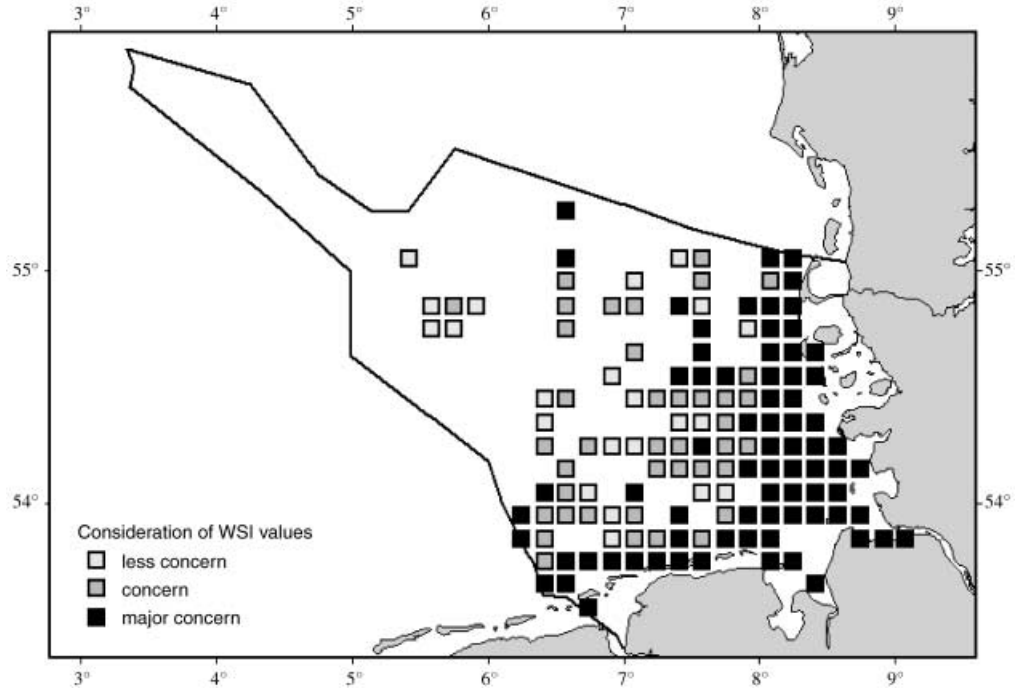


Fig. 6. Areas in the German sector of the North Sea where wind energy utilization is considered to be of 'no (less) concern', 'concern' or 'major concern'. Areas not studied in at least one of the seasons are left blank.

From the average frequency distribution of the WSI values over the four seasons (Fig. 5) it is apparent that most areas do not hold important concentrations of seabirds and thus do not appear particularly vulnerable to marine wind farm construction. However, there is no doubt that some areas have high to very high WSI values and hence are unsuited for such constructions. We suggest a level of concern set at the 60 percentile of the average frequency distribution (= WSI of 24) and a level of major concern set at the 80 percentile (= WSI of 43).

Wind farms operate the whole year round. Threats might therefore also be important even if they only affect species in a single season. Thus, spatial information from Figs 1–4 has been compiled for the whole year. The values of the most important season per grid cell are visualized spatially in Fig. 6 in relation to the levels of concern discussed above.

SYNTHESIS AND APPLICATIONS

This index has been developed primarily for marine wind farm site selection purposes and comparative area assessments. It might be a useful tool for strategic environmental impact assessments (EIA). However, it cannot substitute for proper, detailed EIA, which usually only cover small areas over a limited time period

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(BSH 2003). Results from such small-scale EIA should therefore be set into a more global perspective, provided, for example, by large mapping projects and detailed behavioural studies. Because this is without doubt difficult in normal EIA, particularly in highly dynamic coastal/marine habitats, this study aims to fill this gap by providing comprehensive and up-to-date background information.

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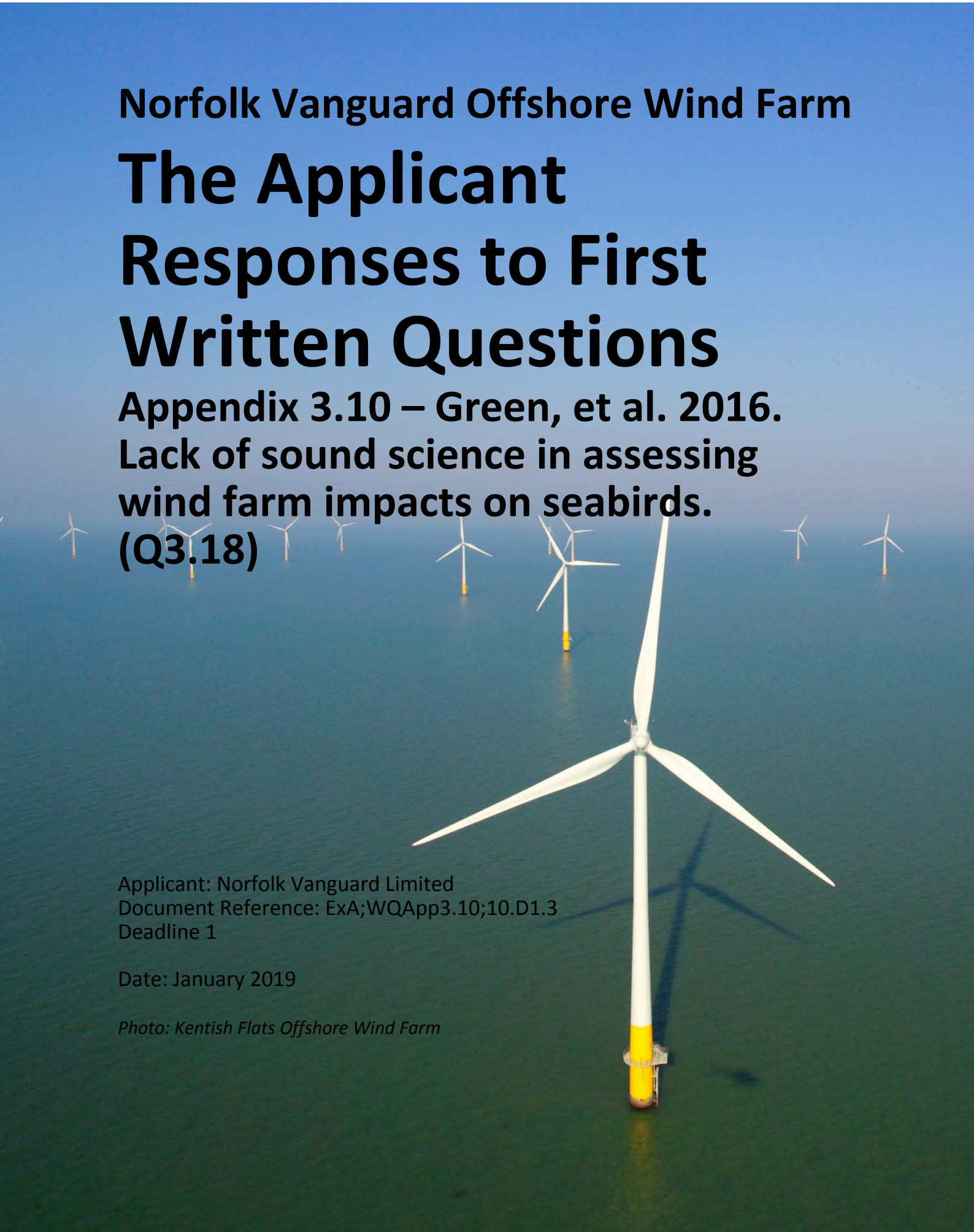
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VATTENFALL

Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.10 – Green, et al. 2016. Lack of sound science in assessing wind farm impacts on seabirds. (Q3.18)



Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.10;10.D1.3
Deadline 1

Date: January 2019

Photo: Kentish Flats Offshore Wind Farm

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PRACTITIONER'S PERSPECTIVE

Lack of sound science in assessing wind farm impacts on seabirds

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Key-words: acceptable biological change, EU Birds Directive, EU Habitats Directive, impact assessment, Leslie matrix model, potential biological removal, site integrity, wind farms, wind turbines

Introduction

Electrical power generation from wind farms has grown rapidly in the UK and European Union (EU) in the last decade and is set to grow further. By 2020, the EU proposes to source 20% of energy from renewable sources (Directive 2009/28/EC). Wind energy is expected to provide 9–14% of global electricity generation by 2050 (IPCC 2011). This may eventually reduce climatic change and its negative impacts on biodiversity, but there are also several poorly quantified negative effects on wild species of renewable energy generation, including wind turbines. For example, birds and bats are killed by colliding with turbine blades or towers and there may be effects of wind farms on mortality and reproductive rates of a wide range of species from avoidance and displacement. Birds may incur additional costs or forego benefits because of reduced transit or foraging within or near to wind farms (Drewitt & Langston 2006; Searle *et al.* 2014). Depending upon the strength of density-dependent compensatory processes, these effects could reduce the population to a lower stable level or cause its extinction (Wade 1998; Niel & Lebreton 2005). Except in the rare circumstances where density dependence is exactly compensating, such effects would always diminish population size. Positive effects of renewable energy infrastructure on populations of wild species have also been proposed and, in a few cases, quantified. These include possible enhancement of food resources of seabirds by protection from fishing from the presence of offshore installations and the provision of artificial substrates as habitat for fish and invertebrates (Inger *et al.* 2009; Langhamer, Wilhelmsson & Engström 2009).

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The UK has the best wind resources in Europe (DECC 2011). Although the cost per megawatt-hour of electricity generation from offshore wind turbines averages about twice that for onshore installations (Bilgili, Yasar & Simsek 2011; Chu & Majumdar 2012), offshore wind power is currently favoured over onshore by the present UK government because of public perceptions of nuisance and landscape consequences of onshore turbines. The UK also has internationally important breeding populations of seabirds. It holds more than 10% of the world's breeding population of eight species, of which three have more than half of their global breeding population in the UK (Brown *et al.* 2015). Because seabirds range over long distances, there may be cumulative impacts on a breeding colony from several wind farms (Masden *et al.* 2010). Seabirds are long-lived and late-maturing, which renders their population growth rate particularly sensitive to additional mortality from collisions or displacement (Niel & Lebreton 2005). The importance of these seabird populations and their sensitivity places a heavy responsibility on those conducting and acting upon scientific assessments of the impacts of offshore wind farms on seabirds to comply with the protection measures and the precautionary principle enshrined in the EU Birds and Habitats Directives (Directive 2009/147/EC and Council Directive 92/43/EEC).

For the UK, and other countries within the European Union, the regulation of wind farm construction requires the assessment of possible damage to the integrity of sites and populations under the EU Habitats and Birds Directives. Consideration must be given to impacts on bird populations of a project on its own and in combination with others already in existence, given consent or planned. Governments give or refuse consent for the construction of wind farms after taking into account the scale and level of certainty of the impacts indicated by these assessments.

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However, there are no definitive quantitative thresholds or criteria defining how large or likely expected impacts must be for damage to the integrity of sites and populations to be anticipated and for consent for wind farm construction to be denied or limited. Consent can be granted only if it is ascertained that there will not be an adverse effect on the integrity of a Natura site, excepting in cases where there are imperative reasons of overriding public interest for consent and no alternative solutions (Article 6 (4) of Directive 92/43/EEC). In recent years, several plans for large offshore wind farms have been approved and some built in UK and EU waters close to large seabird populations because the competent authority judged there was no expected adverse effect on the integrity of the Natura sites involved. For example, in 2014 approval was granted for several extensive wind farms at Hornsea (England, UK Government) and the Firth of Forth (Scotland, Scottish Government), close to internationally important breeding populations of seabirds. This approach contrasts with that in some other EU states. In Germany and Denmark, for example, offshore wind farms have been subject to rigorous marine spatial planning with the aim of avoiding potential conflict with nature conservation as part of the required Strategic Environmental Assessment (SEA) process recommended in EU Commission guidance (European Commission 2011). The German Cabinet approved Europe's first maritime spatial plan in September 2009, after a considerable effort in terms of surveys and research to identify marine sites of high nature value and potential conflict areas with wind farms and to establish zones for various activities and infrastructure. The offshore SEA covering UK waters is not of comparable quality.

In this perspective, we argue that the methods and data used in these cases for estimating effects upon seabird demographic rates and translating them into potential impacts on seabird populations do not allow adequate assessment of effects on site integrity. As a result, sound science and its logical interpretation are lacking in Environmental Impact Assessments of this large and expanding industry.

Estimates of the effects of wind farms on seabird demographic rates are neither robust nor validated

Collision risk models (CRMs) are used to predict the number of fatal collisions of flying birds with wind turbines and per capita additional mortality rates. In the UK, the most widely used CRM is that of Band (2012) (see review by Masden & Cook (2016)). The model requires estimates or assumptions about bird numbers and ages at the wind farm, attribution of birds at the wind farm to source populations, sizes and age structure of source populations, flight behaviour and avoidance rates. Data specific to the project and species being assessed are usually collected on seabird numbers and flight heights, judged by eye, but these estimates are subject to substantial uncertainties, variability and potential biases (Johnston *et al.* 2014), including:

1. accuracy of input variables is rarely quantified, is often poor, and the CRM outputs are highly sensitive to the values used, including flight speed (Masden 2015), and avoidance rate estimates;

2. in many cases, birds at risk are not attributed to source populations because recently developed tracking technologies are either not deployed at all or not on a sufficient scale for robust estimation;

3. count and flight height data are usually insufficient in quantity and quality for precise estimation of seasonal variation, age structure and age differences (Band 2012).

Total avoidance rates used for CRM calculations for seabirds, including within-wind farm avoidance of individual turbines and macro-avoidance by movement of birds around the turbine array, are most often based upon judgement or extrapolation from other contexts rather than pertinent data. Empirical values are only available from a few species (mostly gulls and terns) and usually extrapolated from studies of onshore wind farms, where different circumstances prevail (Cook *et al.* 2014). Robust direct estimates of within-wind farm avoidance rates are lacking for seabird species frequently present in and near planned and consented offshore wind farms in the UK, such as northern gannet *Morus bassanus* and black-legged kittiwake *Rissa tridactyla* (Cook *et al.* 2014). Macro-avoidance and displacement rates have been estimated using radar, visual surveys and imaging, but robust quantitative estimates with confidence intervals are generally not used in impact assessments. Estimates of macro-avoidance for the same species can be highly variable (e.g. Petersen *et al.* 2006; Krijgsveld *et al.* 2011; Vanermen *et al.* 2012, 2013 for northern gannet). This may well be because macro-avoidance varies with the relative positions of nesting and foraging sites, foraging site quality and seasonal timing of studies.

At onshore wind farms, carcasses of some of the birds killed by collisions with turbines can be collected during systematic searches and probabilities of their detection can be estimated. This allows estimation of numbers of deaths per unit time and confidence intervals, even if with low precision (e.g. Bellebaum *et al.* 2013). These methods help to quantify uncertainty and remove bias, but are currently impractical for offshore wind farms. Alternatives that use video or thermal camera systems have not yet been deployed sufficiently to substitute for them. Where direct measurements of avoidance rates are lacking, Band (2012) recommends use of a range of plausible values. However, this can result in a 20-fold variation in assumed per capita mortality rates (APEM 2015).

Overall, CRM outputs are sensitive to the combined effects of multiple assumptions of unknown accuracy, sampling errors and unquantified biases. Only for species that almost completely avoid entering wind farms can the annual per capita mortality rate from collisions be

estimated reliably and with robust confidence limits (Desholm & Kahlert 2005). Validation tests of offshore seabird CRM outputs, in which expectations from pre-consent data and modelling are compared with independent robust post-construction measurements of numbers of collision deaths, have not been conducted.

Estimation of effects on seabird demographic rates of the displacement and barrier effects of wind farms is even less well developed. Avoidance of wind farms by foraging and migrating birds can be substantial and operate over long distances from the turbines (Desholm & Kahlert 2005; Petersen *et al*. 2006; Percival 2010), but the degree to which this affects travel times and costs, access to food and mortality and reproductive rates of breeding seabirds has not been measured reliably. In the case of migrating birds, the displacement and increased travel costs caused by avoidance of a single wind farm may be trivial relative to the total length and cost of the journey (Masden *et al*. 2009), but effects on demographic rates have not been robustly quantified by empirical studies for central-place foraging breeding seabirds repeatedly subjected to barrier or displacement effects. Simulation modelling has been performed of potential effects of displacement by as yet unconstructed wind farms on seabird time and energy budgets and demographic rates (Searle *et al*. 2014). Modelled potential effects of displacement included considerable declines in adult survival of up to 2·1% for black-legged kittiwake and up to 4·9% for Atlantic puffin *Fratercula arctica* (both for the Forth Islands cumulative effects: table 3·3 of Searle *et al*. 2014), though simulated effects on survival for other species and sites and for breeding productivity generally were small. The species for which collision mortality can be reliably estimated as low, because of strong avoidance, are those for which displacement and barrier effects upon demographic rates are potentially the largest, but currently unquantified.

In summary, the procedures currently used to calculate expected effects of proposed wind farms on seabird per capita mortality rates and breeding success largely involve modelling with little firm empirical data. Moreover, actual outcomes at wind farms that have been constructed have not been measured, so model predictions are not tested and there is no adaptive improvement of the decision-making process (Nichols *et al*. 2015). As a result, scientifically robust and defensible calculations of effect sizes for changes in seabird demographic rates caused by collision, displacement and barrier effects of offshore wind farms, with confidence intervals, are currently lacking.

Procedures for translating effects on demographic rates into projected impacts on seabird population size and trends are inappropriate and untested

Assessments of the impacts of offshore wind farms in the UK on seabirds require that the highly uncertain estimates of effects on demographic rates are translated into projections of impacts on population size or trend. Decisions about UK offshore wind farms have been based upon, or influenced by, the following *effect–impact translation procedures*.

POTENTIAL BIOLOGICAL REMOVAL (PBR)

The recommended and robust application of this method is to identify a level of additional mortality *above* which a decline of the affected population to eventual extinction would be likely (Niel & Lebreton 2005). In recent cases, such as Hornsea, the UK statutory conservation agencies advised using this method in wind farm assessments to identify demographic rate thresholds *below* which additional mortality estimated from CRMs and related methods is unlikely to adversely impact the population (Natural England 2014). This reverse application involves faulty logic because PBR's value of maximum potential excess growth may not be realizable in the ecological circumstances of a particular population of interest. In addition, PBR does not estimate the effect of additional mortality on population size.

Potential biological removal provides thresholds of additional mortality that are sensitive to assumptions made about the form of density dependence. The studies of Wade (1998) and Bellebaum *et al*. (2013) show that the shape parameter of the generalized logistic equation has a strong effect on PBR results. Details of the form of density-dependent relationships are rarely known for animal populations and are unknown for any of the UK seabird populations to which PBR has been applied. These uncertainties have prompted the use of 'recovery factors', which are constants by which the maximum possible value of the PBR threshold is multiplied to give a safety margin (Dillingham & Fletcher 2008). The values used for these recovery factors are based upon judgement. There has been no empirical validation of their safety by observation of the effects on population size of known additional mortality rates from any source in any bird species.

ACCEPTABLE BIOLOGICAL CHANGE (ABC)

This method, which has not yet been published in the peer-reviewed scientific literature, was developed by Marine Scotland, a Scottish government agency, and used in a recent assessment of the impact of wind farms on internationally important seabird populations in the Firth of Forth (Marine Scotland, 2015). It uses probabilistic forecasts from stochastic seabird population models to assess the probability of a particular level of population size occurring at some future time, such as the end of the period of operation of a wind farm, in the absence of the wind farm. In practice, this probability is obtained from a simulation model of the population in which variation in expected future population size arises from supposed future demographic and environmental stochasticity in demographic rates, when applied to the population of a

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specified initial size over a period of 25 years, which is the usual licence period for an offshore wind farm. If the best estimate of future population size, after the expected effects of the wind farm on demographic rates are taken into account, equals or exceeds the population size that is 66·7% likely to be equalled or exceeded in the absence of the wind farm, then ABC deems that the impact of the wind farm is acceptable.

The weaknesses of this approach are severe. First, the accuracy of projections of the demographic rates used in the model of the unimpacted seabird population long into the future is highly uncertain and untested. Perversely, the greater the estimated uncertainty, the larger the acceptable population decline. Secondly, it does not address the uncertainties in size of the effects of the wind farm on demographic rates, which are mostly unquantified. Hence, ABC does not assess the risk or probability that the wind farm itself will cause a particular specified outcome or change at all. It simply proposes that an event half as likely to occur as not *if there is no wind farm* should be the threshold for acceptability. Thirdly, the threshold probability for acceptance is arbitrary and is plucked from an unrelated context: IPCC guidelines about the appropriate language to describe the likelihood of an event or outcome of at least given size happening, based upon available evidence (Mastrandrea *et al.* 2010). The threshold chosen for ABC is described as 'unlikely' in the IPCC lexicon. However, this lexicon was not developed for the purpose of determining acceptable levels of risk, which also requires that the societal costs and benefits of possible outcomes are evaluated. It is not only the chance of being wrong that is important, but also the scale of the damage caused by being wrong. No justification is given by the proponents of ABC for using as a tolerable risk threshold for damage to important nature conservation sites and their species a term selected arbitrarily from a lexicon developed by IPCC for a different purpose.

DECLINE PROBABILITY DIFFERENCE (DPD) METHOD

Large uncertainties in predicting future seabird population changes might not matter if *differences* in the probability of a specific population outcome between scenarios with and without wind farms could be predicted reliably and used as criteria for acceptability. This focus on differences in risk has been proposed by the Joint Nature Conservation Committee & Natural England (2012). It was suggested that assessments of acceptable impact should be based upon an arbitrary threshold level of absolute difference between the impacted and unimpacted scenarios in the probability that a population decline by an arbitrary proportion of the initial level would occur. In principle, this approach is preferable to ABC because it takes the uncertainty in the predicted magnitude of the effect of the wind farm into account. However, the results of this procedure are sensitive to the selection of unpredictable baseline (unimpacted) demographic rates. For example, in a model in which the selected values of baseline demographic rates imply a rapid increase in projected population size, it is unlikely that even large additional mortality would give rise to an appreciable absolute difference in the probability of population decline between impacted and unimpacted scenarios. Both probabilities would be very small. If the selected rates were inaccurate and the true values instead led to the unimpacted population being approximately stable, the same level of additional mortality could result in a large difference in the probability of population decline between impacted and unimpacted scenarios.

In practice, uncertainties in future projections of both unimpacted and impacted populations are mostly unquantified, so the probability distribution of an outcome for population size cannot be calculated. This problem makes approaches, such as ABC and DPD, which are based upon assessments of probability or difference in probability unworkable, given present knowledge.

The danger of acceptability thresholds without a logical or empirical basis

All the effect–impact translation procedures described above have a built-in threshold for an acceptable impact. Such thresholds are naturally attractive to decision-makers because they appear to offer a clear-cut, evidence-based way to establish whether damage to the integrity of a designated site will or will not occur. However, in the case of ABC and DPD, the thresholds offer only false security because they are arbitrary, have no foundation in population biology and embed the acceptance of some adverse impact on population size. Whilst PBR does identify a threshold based upon population biology, it is one that is misapplied to the problem at hand. PBR could be used to identify a threshold level of effect of wind farms on demographic rates above which a decline of an affected closed population to eventual extinction would be almost certain. However, population declines of a wide range of magnitudes, short of extirpation, could be caused by effects of wind farms on demographic rates well below this. How large these declines would be depends upon the form and strength of density dependence, which are unlikely to be measured with sufficient precision, and the magnitude of such declines has not been quantified using PBR in any UK wind farm assessment. We argue that such declines would constitute adverse effects on site integrity. Hence, PBR is not an appropriate method for assessing population impacts of a development in a manner that is relevant to the concerns of the public and decision-makers.

A robust effect–impact translation procedure without a built-in threshold

A more robust procedure for evaluating population-level impacts of wind farms on seabirds is to calculate, using a

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density-independent Leslie matrix model (LMM), expected population sizes, with and without the expected effects on demographic rates of the wind farm, at the end of its lifetime. The ratio of the expected population size with the wind farm to that without it (the counterfactual of population size) is a robust metric for likely population-level impact of a specified set of effects of the wind farm on seabird demographic rates. This LMM-ratio approach is relatively insensitive to the assumptions made about the magnitude, variability and trends of demographic rates in the model from which it is calculated, because the same uncertainties apply to both the impacted and unimpacted scenarios. Hence, this effect–impact translation procedure contributes little to the uncertainty in the difference in population size caused by the wind farm.

Density dependence tends to reduce the impact on population size of a given effect of the wind farm on demographic rates, so the LMM ratio calculated from the density-independent model is a precautionary worst-case outcome. We think it probable that density-dependent compensation occurs in UK seabird populations and that including it in LMMs (e.g. Miller, Jensen & Hammill 2002) could lead to more accurate estimates of population impact than those based upon density-independent LMMs. However, accuracy would only be increased if robust estimates of the form and strength of density dependence were available or population outcomes could be shown to be insensitive to assumptions made about density dependence in the absence of reliable quantification. In practice, no assessments of population impacts of additional mortality from wind farms on UK seabirds have included empirical estimates of the form and strength of density dependence because applicable estimates seem not to be available. Until adequate quantification of density dependence is available, we recommend the use of density-independent LMM ratios.

Whether density dependence is included or not, there is no threshold value of acceptability built into the LMM-ratio metric. Population estimates from Leslie matrix models, for example Trinder (2014), and population models fitted using a Bayesian approach (Marine Scotland, 2015) have been calculated as part of offshore wind farm impact assessments, but their results have not been used explicitly as counterfactuals in decision-making about the acceptability or otherwise of UK offshore wind farm projects. Based upon the documentation of UK wind farm assessments, we believe that methods such as PBR, ABC and DPD have been used in preference to LMMs because they provide thresholds which can be used to argue that site integrity will not be affected by the project, whilst LMMs deliberately do not provide a threshold. We argue that, because the thresholds offered by the other methods are arbitrary and invalid, LMMs should be used as the standard, best-practice method, and we note that any of the potential positive effects of offshore wind farms on seabird demographic rates, if quantified, could be included in an integrated assessment using an LMM-ratio metric.

Conclusions

Current procedures for collecting empirical data, modelling effects on demographic rates and translating those effects into projected impacts of offshore wind farms on seabird populations are inadequate. Empirical measurements of effects of offshore wind farms on seabird demographic rates from fieldwork are not sufficiently precise and unbiased. In the case of some important parameters such as turbine avoidance rates and the strength of density-dependent compensation, estimation is rarely even attempted. As a result of these holes in the evidence base, the magnitude of effects of wind farms on seabird demographic rates cannot be estimated accurately and the level of bias and precision in the estimates used cannot be calculated.

To overcome these problems, responsible governments should require the renewable energy industry to co-fund an adequate level of field-based research to estimate effects of wind farms on seabird demographic rates more reliably. The Offshore Renewables Joint Industry Partnership (ORJIP) intends to address this need (Carbon Trust 2015), but the objectives of its project need to be greatly expanded with regard to the number of species covered, proximity to their breeding colonies and robustness of estimation. Further development and deployment of radar, imaging and tracking techniques are likely to be required, including remote download 3D tracking (Cleasby *et al.* 2015). A defensible approach is then needed to translate these effect measurements, and their uncertainties, into expected impacts on populations. We propose that the counterfactual population ratio from a density-independent Leslie matrix model would be an appropriate method for this translation.

Quite separate from these problems of measurement, estimation and modelling, there is a fundamental logical flaw in the link between scientific assessment and decision-making about the acceptability of wind farm impacts. Modelling approaches have been contrived that seek to define an acceptable threshold for a projected negative impact of a wind farm on seabird populations, below which this negative impact is regarded as causing no adverse effect on site integrity. However, the emperor has no clothes: the thresholds used to define the acceptability of projected offshore wind farm impacts are arbitrary, poorly reasoned, not designed for the purpose and have no valid biological basis. Hence, it is necessary to revise decision-making procedures, regardless of what effect-to-impacts translation procedure is used. At present, inadequate data are being combined with arbitrary and scientifically unsupportable thresholds to argue that wind farms will cause no damage to the integrity of sites designated to restore and maintain Europe's biodiversity.

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Population viability analysis indicates that the probability of long-term persistence of an animal population and its mean time to extinction generally increases with its average size (Akçakaya, Burgman & Ginzburg 1999). According to European Commission guidance on managing the network of protected sites established by the EU Birds and Habitats Directives (Natura 2000 sites), Article 6 of the Habitats Directive provides that 'The integrity of the site involves its ecological functions. The decision as to whether it is adversely affected should focus on and be limited to the site's conservation objectives' (European Commission 2000). In addition, for the integrity of a site not to be adversely affected, a Court of Justice of the European Union decision (Court of Justice of the European Union 2013; Para. 39) found that the 'site needs to be preserved at a favourable conservation status', which entails 'the lasting preservation of the constitutive characteristics of the site concerned that are connected to the presence of a natural habitat type whose preservation was the objective justifying the designation of that site'. Based upon this reasoning, we argue that some damage to the integrity of a designated site will have been sustained if populations of the seabirds for which it was designated are diminished, even to a small degree, by the effects of a wind farm, compared with what they would otherwise have been. If that is expected to be the case, it does not mean that the competent authority cannot give consent for a wind farm. Article 6(4) of the Habitats Directive sets out tests that determine whether the expected damage can be accepted and compensated for. However, poor science should not be used to avoid those tests by claiming that no damage will occur.

Data accessibility

Data have not been archived because this article does not contain data.

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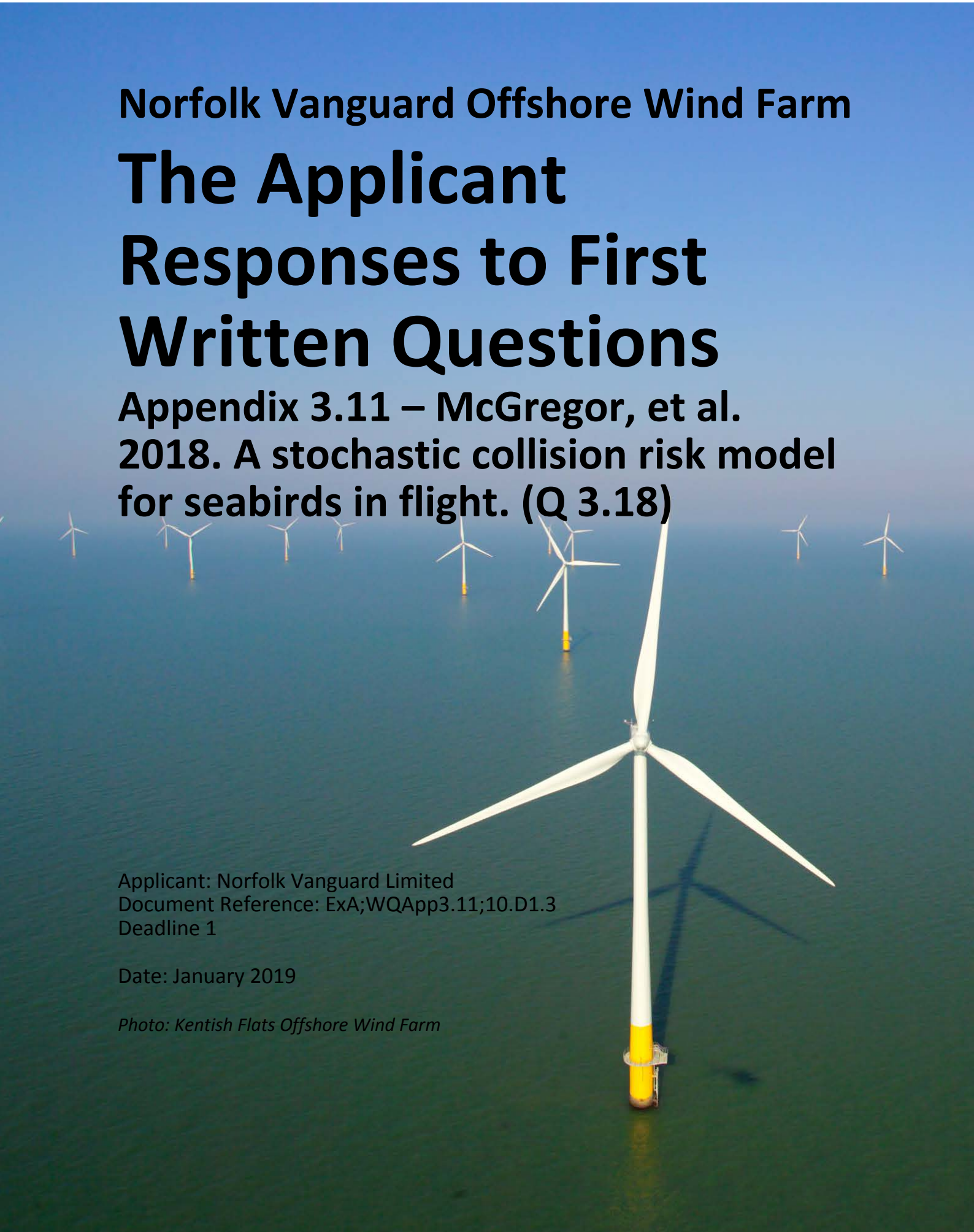
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The Applicant Responses to First Written Questions

Appendix 3.11 – McGregor, et al. 2018. A stochastic collision risk model for seabirds in flight. (Q 3.18)



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A Stochastic Collision Risk Model for Seabirds in Flight



A Stochastic Collision Risk Model for Seabirds in Flight

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Executive summary

1 Collision Risk Models (CRM) are used to assess impacts on seabird populations in all offshore wind farms Environmental Impact Assessments ('EIA') and Habitats Regulations Appraisals ('HRA') in the UK. Existing models are unable to properly incorporate uncertainty in the input parameters into calculations of uncertainty in the collision prediction and consequently are not expressed in the outputs.

2 Uncertainty in predicted collision has resulted in the delayed deployment of offshore wind projects, with projects being reduced in size or even cancelled. Not incorporating uncertainty when it is known to occur may be failing to meet the requirement from the European Court of justice to use, "…the best scientific knowledge in the field…".

3 This project aimed to create a CRM that incorporates variability in input parameters correctly into a predicted collision impact with estimated variability. In order to produce a model that was fit for purpose, stakeholders were consulted through a questionnaire-based survey.

4 The survey results section was in seven parts, each asking about different aspects of the CRM. These were: CRM concept, user experience, CRM inputs, CRM operation, CRM outputs, CRM error checking and CRM improvements.

5 The survey, while taking in to account the scope of the project, resulted in the following changes requested by stakeholders:

- Create a user-friendly interface for non-R users;
- Speed up the code;
- The number of turbines should be a user input;
- Output predicted collision probability data;
- Seasonal (as well as monthly & annual) assessment (default + user defined);
- Error checking inputs and collision probability; and,
- Monthly or seasonal flight height inputs.

6 The new stochastic CRM (sCRM) was based on the code written by Masden (2015), but had to be compatible with the Band (2012) offshore CRM. Testing showed that the predictions of the Masden (2015) code matched the predictions of the Band (2012) Excel spreadsheets for Option 1, but that differences in outputs for Options 2 and 3 arose because of a calculation error in Masden (2015) code. Consequently, the sCRM was based on an updated, and streamlined, version of the Masden (2015) code.

The new sCRM was produced in two forms: Firstly, a Shiny app based on the R-code, available as an online tool, which can be run from:

https://dmpstats.shinyapps.io/avian_stochcrm/

Secondly, the Shiny app can be downloaded as a package and run locally in a browser. It can be downloaded from:

https://github.com/dmpstats/stochCRM

1 Introduction

7 Collision Risk Models ('CRMs') have been used to assess impacts on bird populations in all offshore wind farms Environmental Impact Assessments ('EIA') and Habitats Regulations Appraisals ('HRA') in the UK since 2009. These types of models have also been used in onshore wind farm EIA and HRA since the early 2000s, with further models being produced since then to address various issues (Masden & Cook 2016). They have become a *de facto* requirement of Environmental Statements and Appropriate Assessments ('AA') in the United Kingdom.

8 CRMs as an impact assessment tool began with the production of the Scottish Natural Heritage ('SNH') (Band 2000, Band *et al*. 2007) model, which is an application of the concept first published by Tucker (1996). It is a simple mechanical model that calculates the probability of a bird of a certain size moving at a set speed through a wind turbine rotor, being struck by a turbine blade of a certain size and moving at a set speed. Since it is a simple mechanical model of two bodies in motion it does not account for bird behaviour in avoiding the wind farm, or a turbine or the rotor blade itself. These elements of bird behaviour (as well as any errors in the calculation) should, hypothetically, be taken into account by applying an avoidance rate (typically 95% or higher). The Band (2000) model was designed for onshore wind farms where data on bird flight activity is collected by observers carrying out behavioural observations prior to the wind farm be constructed. However, the data required to characterise the ornithological interest in an offshore wind farm makes use of very different data. Boat based or digital aerial surveys are undertaken to estimate species density. It was therefore necessary to adapt the SNH (2000) model to use this type of data.

9 This was undertaken by Bill Band (the original author of the SNH (2000) model), for The Crown Estate Strategic Ornithological Support Services ('SOSS'), under the Round 3 enabling actions. This new model, like the SNH (2000) model, was provided in Microsoft Excel spreadsheets and was deterministic (Band 2012). While the guidance to the Band (2012) model did suggest an approach to incorporate variation around input data, the method was not statistically valid as it assumed that each variable was independent (Masden 2015), and there were errors in the assumed levels of variability around some input values.

10 The limitation of the Band (2012) model in incorporating input value variability and uncertainty led to Masden (2015) developing a stochastic version of the Band (2012) model. In addition to incorporating data uncertainty in to the model, the Masden (2015) version also coded the calculations in to R code (http://www.r-project.org). However, while Masden (2015) successfully achieved the coding of the Band (2012) model and incorporating uncertainty, users have noted various flaws in running this code. This culminated in a review of the Masden (2015) version of the model by Trinder (2017).

11 The main findings of Trinder (2017) were that the Masden (2015) coded version of Band (2012) has the following constraints:

- The use of only normal distributions or truncated normal distribution for all variables was inappropriate;
- Turbine parameters are modelled with uncertainty, which does not meet the requirement to follow a 'Rochdale envelope' approach to consenting;
- The Masden (2015) code did not allow bird aerial densities to exceed two birds per km^2, which was unrealistic;

- The model *always* uses the generic wind speed, rotor speed, blade pitch relationship provided, and this cannot be 'switched off'; and,
- The method used to generate a range of proportions of birds at collision risk height *can* generate negative values.

12 In most circumstances, the deterministic outputs from the SOSS CRM have been sufficient for determining no likely significant effect on the environment, for EIA, or no adverse effect on site integrity, for an AA. In most cases, for most species, it can be clear that, even with a worst-case scenario used as input parameters, the predicted impacts are relatively small. Uncertainty in CRM can have large impacts on the deployment of offshore wind projects; e.g. the Docking Shoal project was refused consent in July 2012 based on the outputs of CRM, and subsequent population modelling, and it is therefore essential that models are able to be relied upon by developers, regulators and advisers. As the number of developments increases this will be applied increasingly via cumulative impact assessments.

13 However, there have been increasingly frequent situations where CRM predictions have come very close to significant impacts. In these situations, an over-reliance on a single-value CRM prediction can lead to problems, even when a worst-case scenario is presented. Thus, an understanding of the variability around input values and their effects on the potential range of output values can be very important. Existing case law suggests that the approach using a single, precautionary, value may not be wholly compatible with the purpose of the European nature directives.

14 The European Court of Justice ('ECJ') Case C-127/02 states that an appropriate assessment should be made, "…in light of the best scientific knowledge in the field.". It could be argued that a deterministic CRM is not making use of the "best scientific knowledge" as it is known that input values are variable, and the only approach to use in these situations is potentially unrealistic worst-case scenarios. A stochastic CRM would not have these problems, as it would incorporate the variability in the data and present a result with levels of uncertainty. Thus, worst case scenarios can be avoided and the best scientific knowledge in the field can be used appropriately. Outputs from a stochastic CRM can then be used as a mortality input, with known variability, for stochastic population models. These can be used for predicting the importance of the impact on populations for either EIA or HRA.

2 Aims of this project

15 The research aim of this project was to develop a stochastic version of the Band (2012) collision risk model in R that would incorporate the gaps identified by industry and statutory agencies, providing a more robust and transparent method of accounting for uncertainty in the estimation of seabird collision rates.

2.1 Objectives

16 The research objectives for this project were:

- Identify current gaps in Band (2012) model and Masden (2015) code to be addressed in an R-based stochastic version.
- Produce an R-based stochastic version of Band model, tested against the existing Excel version, with R code independently validated.
- Provide advice on the most appropriate parameterisation of the model produced, accounting for limited information that may be available for some variables and the rapidly evolving wind turbine generator technologies.
- Consider end-users' needs and ensure that outputs presented from the model were in an appropriate form.

3 Stakeholder engagement

17 Positive stakeholder management and consultation is the identification, analysis, planning and implementation of actions to allow clear and open engagement with stakeholders. In this instance stakeholders were individuals or groups with an interest in the project, 'A stochastic collision risk model for seabirds in flight', because they are involved in work on this topic or may be affected by the outcomes from the consultation process.

18 Stakeholder management, and management of aspirations there-in, is a challenging aspect with any consultation. The overall project can be undermined if there are significant areas of confusion with poor stakeholder commitment and a lack of clear engagement, emphasising the need for clear documented communication.

19 The final draft pro-forma questionnaire was therefore fully discussed with the Project Steering Group ('PSG') prior to distribution, with several changes being made.

3.1 Questionnaire

20 A stakeholder questionnaire was designed to capture responses on all the current CRM inputs and outputs, where there are limitations and how stakeholders think these should be addressed. Questionnaires were provided as PDF forms (see Appendix 1), that could be printed and completed by hand or electronically, or via an online survey using Google Forms. Stakeholder responses were also followed up with a telephone interview for a cross-section of stakeholders (Appendix 2).

21 Responses were analysed using descriptive statistics and qualitative analysis, to determine the gaps in existing CRMs and stakeholder needs.

22 Data collected from respondents was anonymised and analysed to determine the key changes needed to be made to the current CRM.

23 Analysis of pro-forma data involved quantitative descriptive statistics and qualitative analysis of free text responses. This included analysis of the response rate, most important concerns about input data, most common concerns about outputs and the most common requested changes to the CRM. These were analysed as a whole for all respondents. Free text was summarised and descriptive assessment of common themes undertaking using word clouds.

24 In addition to the questionnaire a selection of stakeholders were invited to participate in a follow up interview by telephone. This was to ensure that the questionnaire was capturing all of the responses from stakeholders necessary to identify the needed improvements in a stochastic CRM.

3.1 Survey results

25 Survey results were split into seven sections, each asking about different aspects of the CRM. These sections were:

- CRM concept;
- User experience;
- CRM inputs;
- CRM operation;
- CRM outputs;

- CRM error checking; and,
- CRM improvements.

3.1.1 CRM concept

26 There was only one question, Question 1, in this section.

27 Question 1 was in two parts. The first part of the question, 1a, asked, "Do you think that CRM is a useful method for assessing potential impacts from offshore wind farms?"

28 This question was to determine if stakeholders thought that collision risk modelling was a useful method when used for impact assessments. In addition, it provided important context to a stakeholder's views that could affect their responses to other questions.

29 Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

30 All responses were "Yes", though two responses provided qualification on their response. One stakeholder noted that there was too much emphasis on CRM results and that they tended to be taken too "literally". The other response was similar, noting that the value of CRM output depends on how they are used; if as an absolute measure of risk to birds, CRM was not considered useful, but as a relative measure it was considered useful.

31 The second part of question 1 was a free text option, "If you answered "Yes" to Question 1a, please describe the benefits of CRM. If you answered "No", please describe why you think that CRM is not a useful method."

32 Most responses were positive (56%) and were mostly in relation to the existing CRM being quantitative, transparent and consistently applied. Many positive responses highlighted the CRMs value in providing relative impact between turbine scenarios or between projects. Its value as a cumulative impact tool was also mentioned several times.

33 A large proportion of responses (40%) provided comments containing both positive and negative comments. Negative comments were focused on issues around too much use of absolute, rather than relative, impact calculations. Many stakeholders were concerned that CRM outputs tended to be considered as more accurate a measure than the input data suggest. Only one comment (4%) was wholly negative.

34 Analysis using a word cloud (Figure 1) highlights that responses were not entirely positive or negative. The words "provides" and "potential" were common, as were "data", "impacts" and "risk". This matches the findings that more comments were positive, and that they were focused on CRM being useful for assessing potential impacts on birds.



Figure 1 Word cloud analysis results of responses to Question 1b.

3.1.2 User experience

35 There were four questions in the section on user experience. This was split between questions about experience using the Band (2012) and Masden (2015) models, and general use of R-code.

36 Question 2 was also in two parts. The first part of the question, 2a, asked, "How would you describe your primary role in using the Band (2012) CRM for offshore wind farms? (Tick both user and interpreter boxes if appropriate)"

37 This question was asked to determine stakeholders' level of understanding of collision risk modelling and their ability to knowledgably answer questions or provide feedback.

38 Stakeholders were provided with two possible responses

- Model user; and,
- Model output interpretation.

39 Valid responses were either of these options or both. The field was not mandatory, so users could provide no response. Stakeholders were then given further options depending on which of the above options they chose. For model users, there were four possible responses:

- Expert;
- Occasional;
- Basic; and,
- None.

40 For model output interpretation, there were three possible responses:

- Supervisory;

- Reviewer; and,
- None.

41 Those that chose "None" were asked to describe their use of the Band (2012) model. Most stakeholders described themselves as both model users and model output interpreters (Figure 2). There were slightly more stakeholders that described themselves as only undertaking model output interpretation (20%), than only model use (12%). Only 2 stakeholders (8%) did not provide a response.

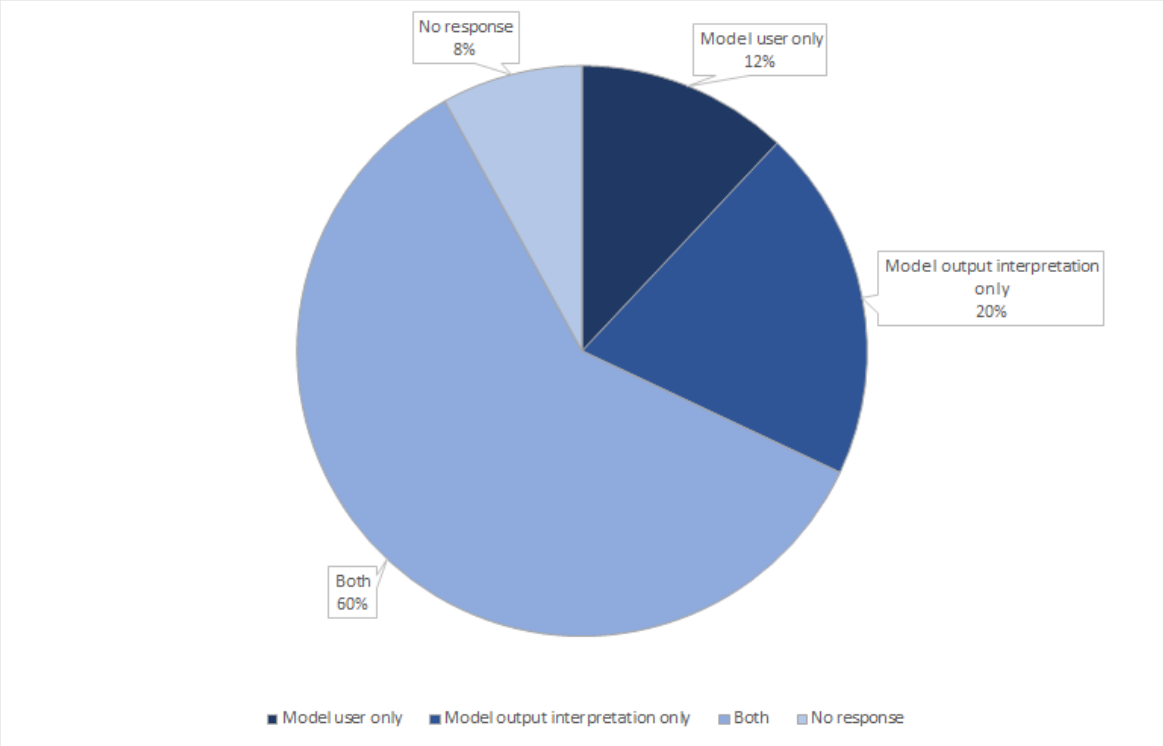


Figure 2 Relative proportion of stakeholder responses to question 2a, Part I.

42 The responses to this part of Question 2 indicated that most stakeholders responding to the survey were well aware of the Band (2012) CRM in some capacity and were therefore likely to provide useful feedback.

43 Among those that described themselves as model users, the majority (46%) described themselves as "Expert" users (Figure 3). Small proportions described themselves as "occasional" or "basic". A relatively large proportion (25%) did not provide a response, but these were mostly stakeholders that described their experience as only with model output interpretation.

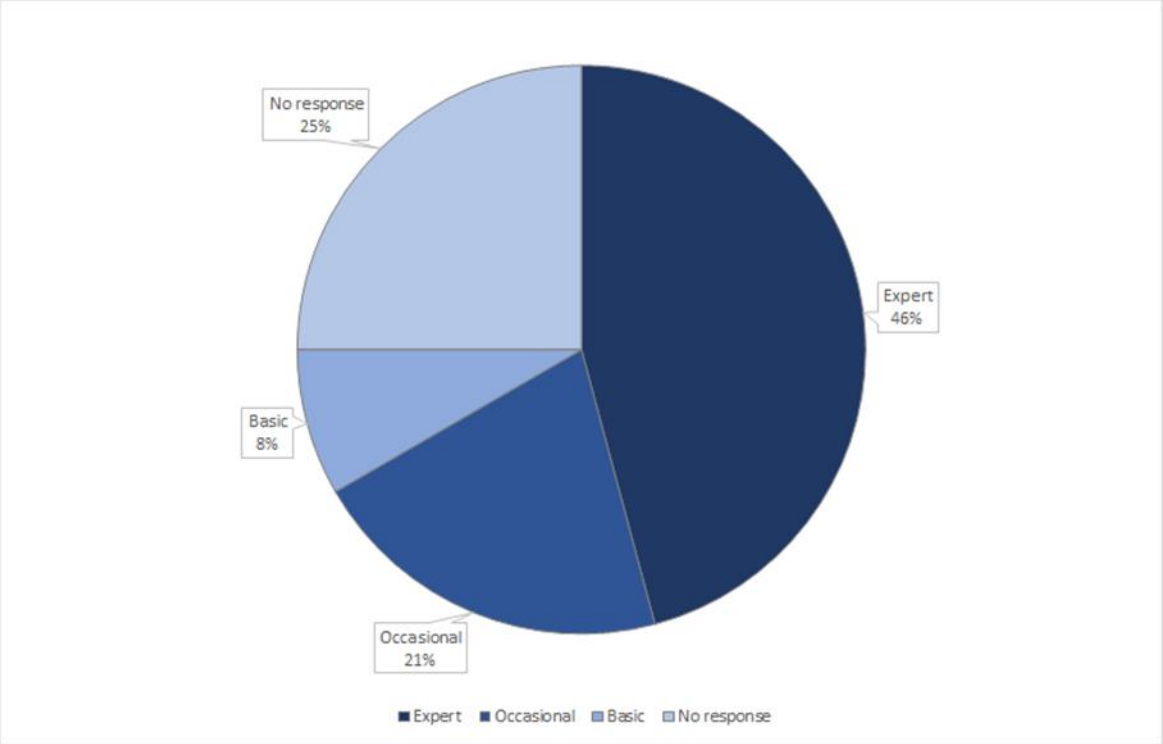


Figure 3 Relative proportion of stakeholder responses to question 2a, Part II.

44 Of those stakeholders that described themselves as being involved with model output interpretation, almost half (48%) were reviewers only of model outputs (Figure 4). Almost one third (28%) were either only supervising model output interpretation or were involved in both reviewing and supervising model output interpretation. Three stakeholders provided the response "other", and three did not provide a response, but these had not selected "model output interpretation" as a response. The free text

responses from three stakeholders only provided confirmation of their status from the categorical responses, so did not provide any further relevant information.

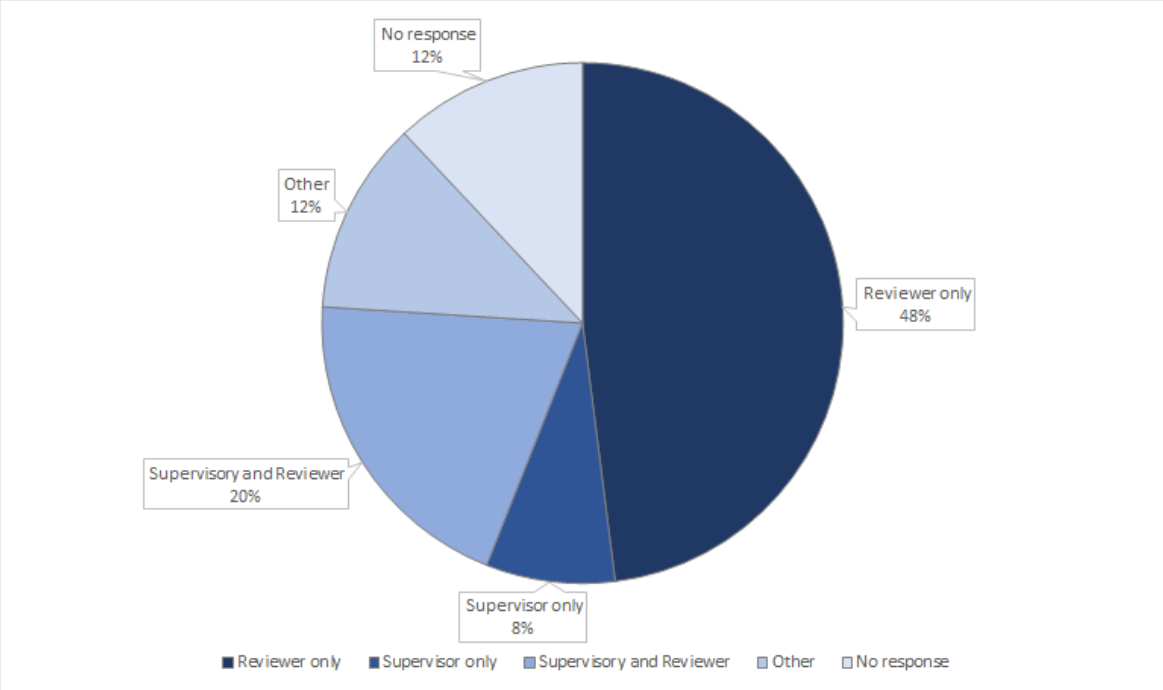


Figure 4 Relative proportion of stakeholder responses to question 2b.

45 Question 3 was a single part question, "What level of R user do you consider yourself to be?". This question aimed to determine stakeholders' level of understanding of the coding language to be used for the stochastic CRM and their ability to knowledgably answer questions or provide feedback.

46 Stakeholders were provided with five possible responses:

- Expert;
- Regular;
- Occasional;
- Never; and,
- Other.

47 Those that chose "other" were asked to provide further information in a free text box. The most common response from stakeholders was that they had no experience of using R (44%), with a relatively high proportion only using it occasionally (24%) (Figure 5). Almost a quarter of responses (24%) were from stakeholders that described themselves as expert or regular users of R.

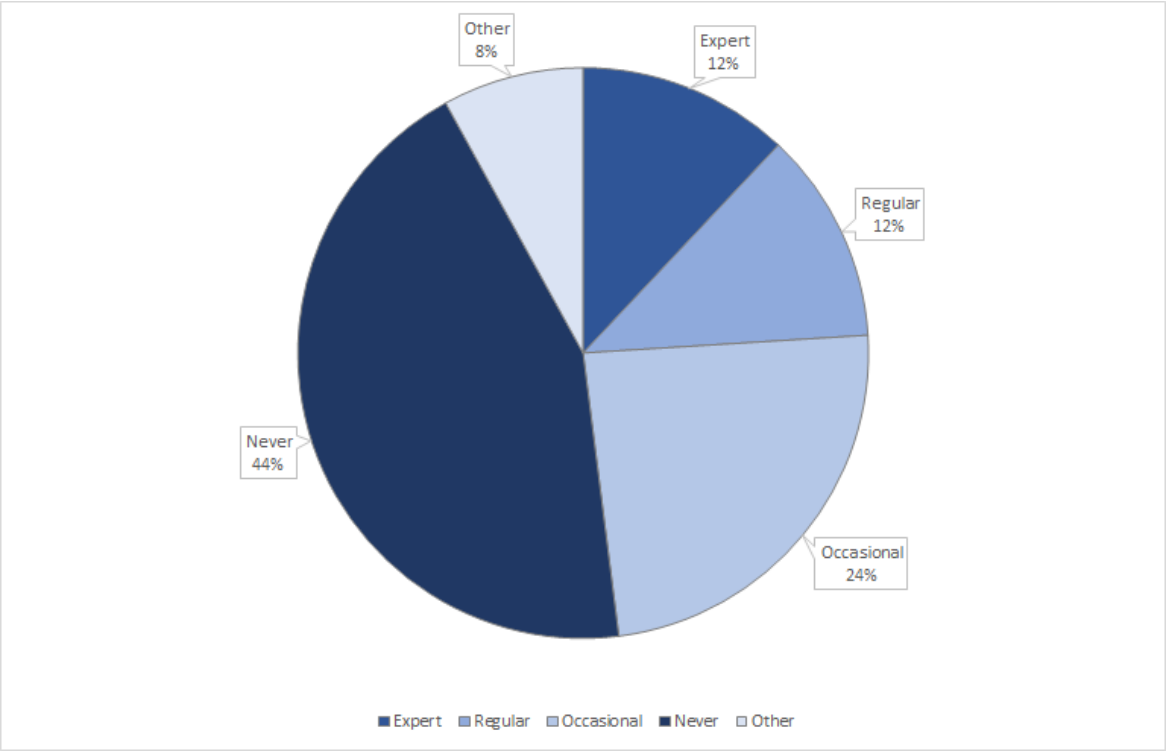


Figure 5 Relative proportion of stakeholder responses to question 3.

48 This made it clear that most stakeholders that responded were unlikely to make a lot of use of an R-code only version of a new stochastic CRM.

49 Question 4 was also a single question, "Have you ever used the Masden (2015) stochastic CRM (or another stochastic CRM) in R?"

50 This question was also to determine stakeholders' level of understanding of CRMs in R, rather than only in Excel, and their ability to knowledgably answer questions or provide feedback. Stakeholders' were provided with four possible responses:

- Yes (Masden (2015) CRM);
- Yes (another stochastic CRM);
- No; and,
- Other.

51 Responses were divided between a majority (60%) that had never used the Masden (2015) CRM, and a large minority (40%) that had. No stakeholders had used any other stochastic CRM, and there were no "other" responses (Figure 6).

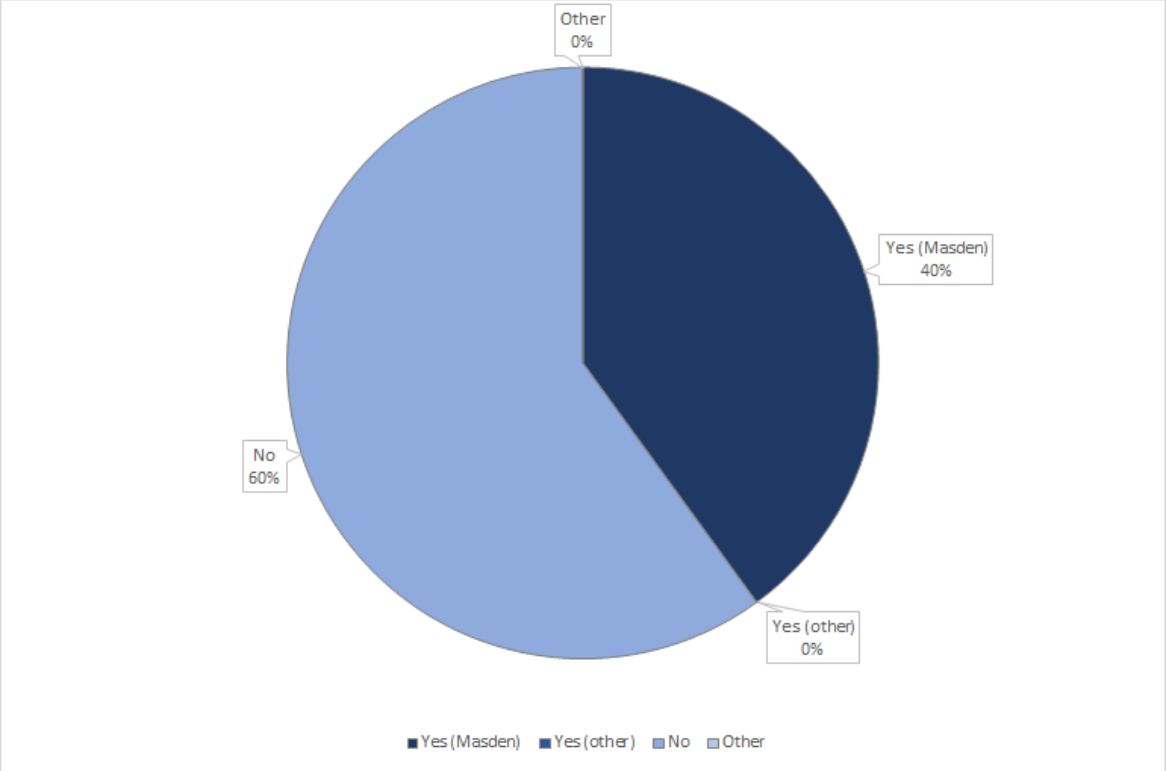


Figure 6 Relative proportion of stakeholder responses to question 4.

52 Question 5 was also a single question, "Have you ever experienced issues running the Masden (2015) stochastic CRM (or another stochastic CRM) in R?" This question aimed to draw out any currently unknown problems with the Masden (2015) version of the CRM.

53 Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

54 A free text box was provided asking those who responded "Yes" to provide further information.

55 While the majority of responses (Figure 7) were either "Don't know" or "No response" (36% and 28% respectively), most responders with a known response had experienced problems with the Masden (2015) version of the CRM (28%).

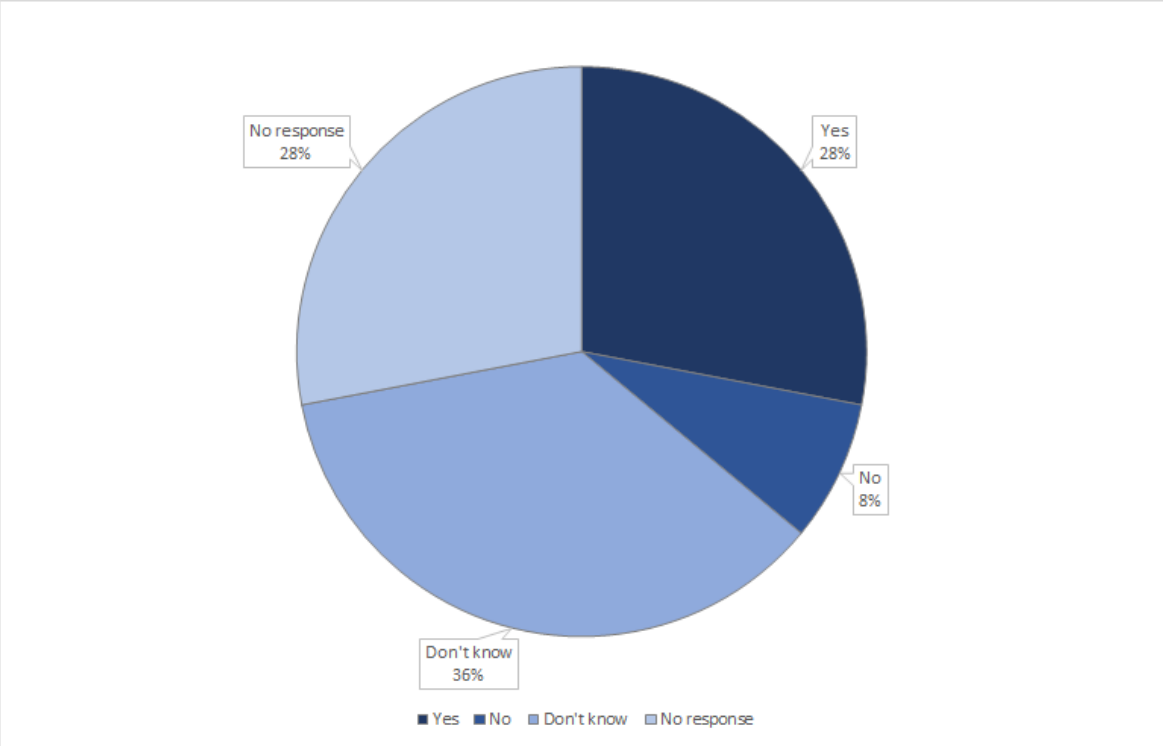


Figure 7 Relative proportion of stakeholder responses to question 5

56 Free text responses were often in relation to bugs in the code, the probability distributions used for count data, the way that the number of turbines is calculated, the assumed relationship between wind speed, rotor speed and blade pitch and the speed to run the model. Useful other comments included issues with selecting appropriate proportions at collision height, variation being present of fixed parameters (e.g. blade length will effectively have no variation around it) and the difficulty experienced when trying to run multiple turbine parameters.

57 Word cloud analysis (Figure 8) of the free text responses agreed with the above assessment with "code", "input" and "parameters", and "problems" being commonly expressed.

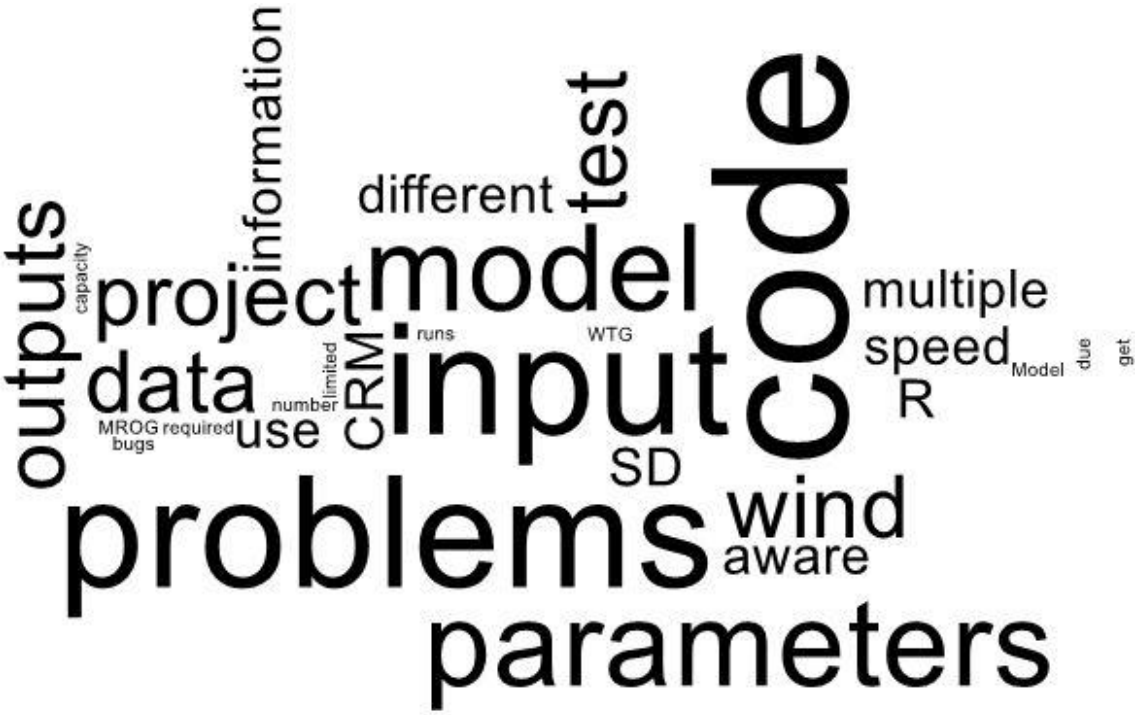


Figure 8 Word cloud of the free text component of question 5

3.1.3 CRM inputs

58 There was only one question in the section on CRM inputs, Question 6.

59 Question 6 was also a single part question, "Are there any Band (2012) input values for birds (e.g. wing span, length, flight speed, nocturnal activity) that you think should be changed, improved or added?"

60 This question aimed to ensure that as many improvements as possible were included in the new model. Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

61 A free text box was provided asking those who responded "Yes" to provide further information. There was a strong, positive, response from stakeholders (76%) to this question (Figure 9). With only 12% stating that there were no changes needed to the bird input parameters.

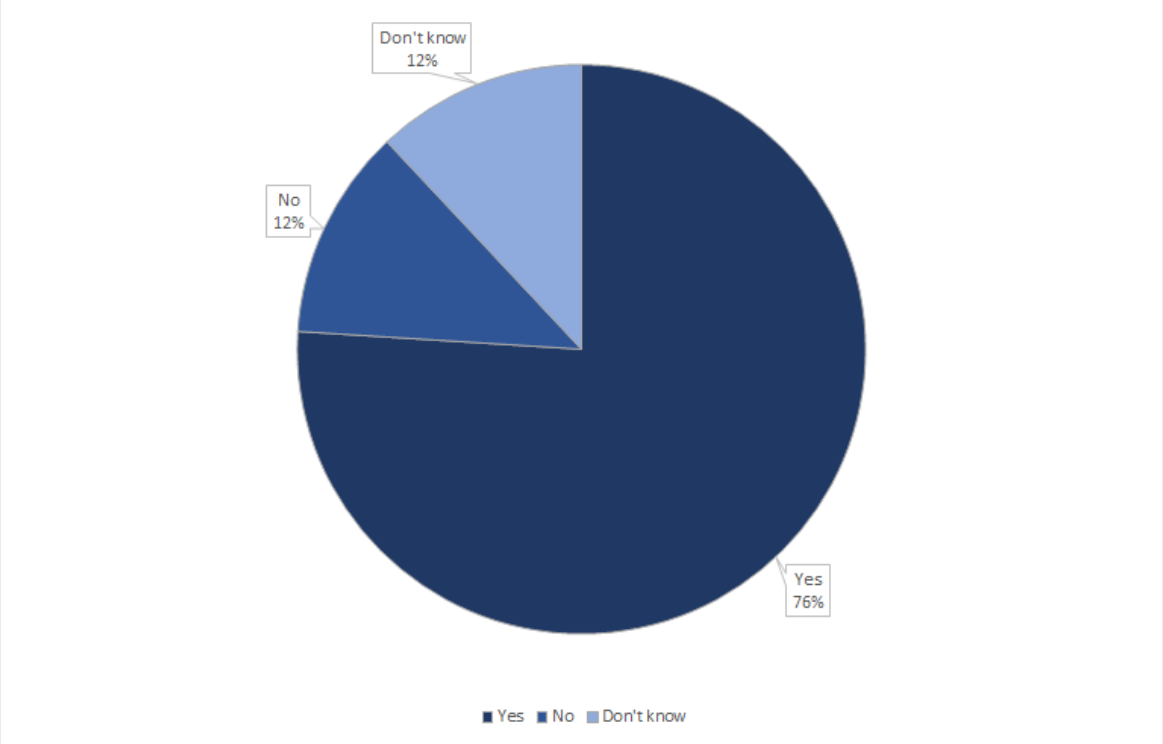


Figure 9 Relative proportion of stakeholder responses to question 6

62 The free text responses from those that stated "Yes" were commonly in relation to flight speed data and nocturnal activity data. Responses suggested that existing data were of poor quality (small sample sizes) or poor resolution (broad categories for nocturnal activity) or both. Other useful comments centred around the lack of behavioural responses in the model (e.g. changes in bird speed, height, etc. in relation to weather). There were also comments that the model is unrealistic in dismissing the effect of different angles of approach to the rotor, though one stakeholder commented that this was not really a bird input parameter issue, but a model calculation issue.

63 Word cloud analysis confirmed much of the above assessment, with "flight", "values", "bird" and "nocturnal" the commonest words used. "Activity", "speed", "model" and "data" were also commonly used.

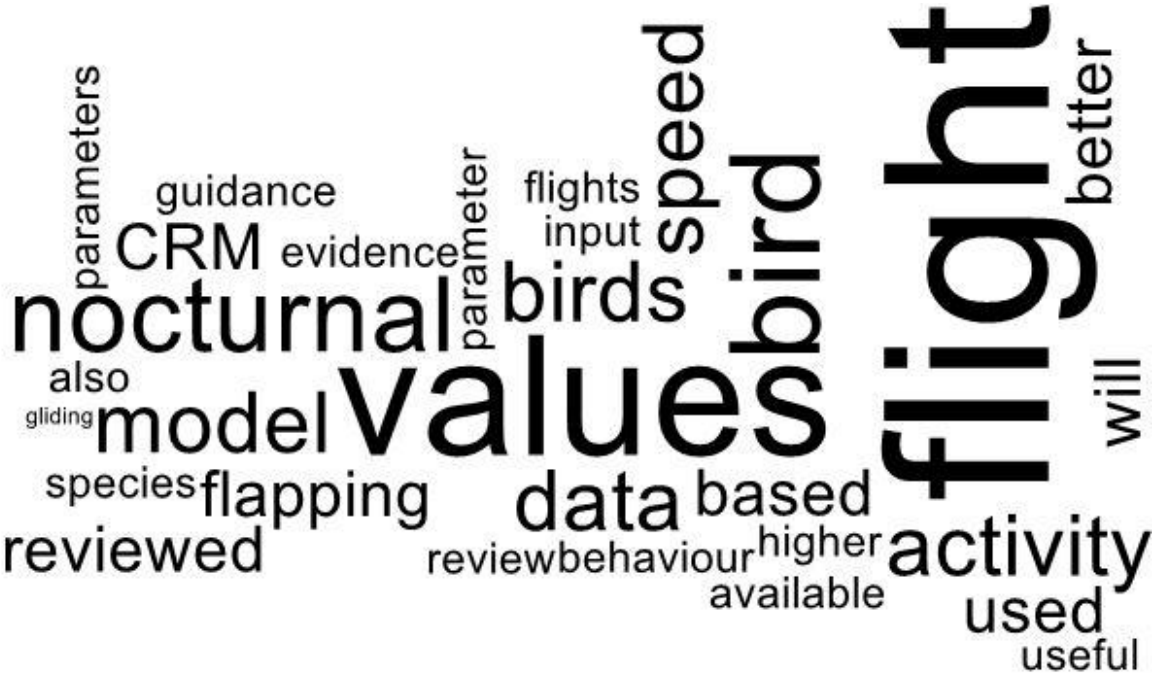


Figure 10 Word cloud of the free text component of question 6

3.1.4 CRM operation

64 There were three questions in the section on CRM operation. These were related to how the CRM does, or should, predict the number of collisions.

65 Question 7 was also a single part question, "Should the new stochastic CRM retain all of the model Options (1, 2, 3 & 4) described by Band (2012)?"

66 This question aimed to gauge whether stakeholders wish to see changes in the approach used for modelling the different options. Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

67 A free text box was provided asking those who responded "No" to provide further information. There was a clear response from stakeholders, with 64% wanting to retain the four model Options available in the Band (2012) CRM (Figure 11). Roughly the same number of stakeholders responded "No" as "Don't know".

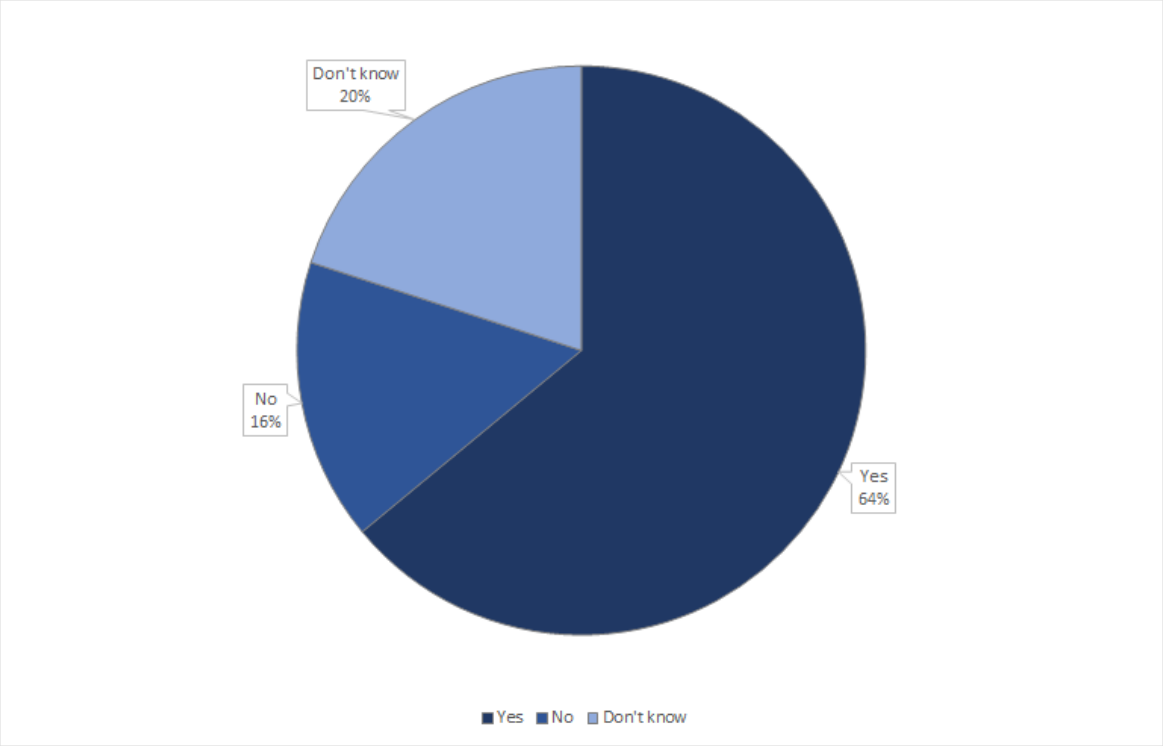


Figure 11 Relative proportion of stakeholder responses to question 7

68 While the questionnaire asked for further information only if the stakeholder responded "No", two of the six responses were from stakeholders who responded "Yes". Both responses noted that all options should be retained for making comparisons with older assessments, so these responses were still very useful. There was no consistent response from stakeholders, with some wanting to drop Option 3 & 4 (extended model), and some wanting only Options 1 & 3. One comment was that if the model is to be stochastic, then only the extended model should be used, as this is the most realistic calculation, as it takes into account the skewed flight height distribution of most seabirds.

69 Question 8 was also a single part question, "The Masden (2015) CRM includes the relationship between wind speed, rotor speed and blade pitch. Given the commercial sensitivity of this information, should a precautionary generic approach be used or should turbine specific data be used for consent applications?"

70 There has been criticism of this approach (particularly the access to suitable turbine data at a pre-consent phase). So was considered important to ask the wider community of stakeholders the implications of either not including this approach, or the potential uncertainties in using generic data.

71 Stakeholders were provided with four possible responses:

- Precautionary generic approach;
- Turbine specific approach;
- Don't know; and,
- Other.

72 A free text box was provided asking for any further information on why the stakeholder gave the response they did.

73 There was roughly an equal split between "precautionary generic approach", "turbine specific approach" and "other". A relatively small proportion (8%) of stakeholders responded "don't' know" (Figure 12).

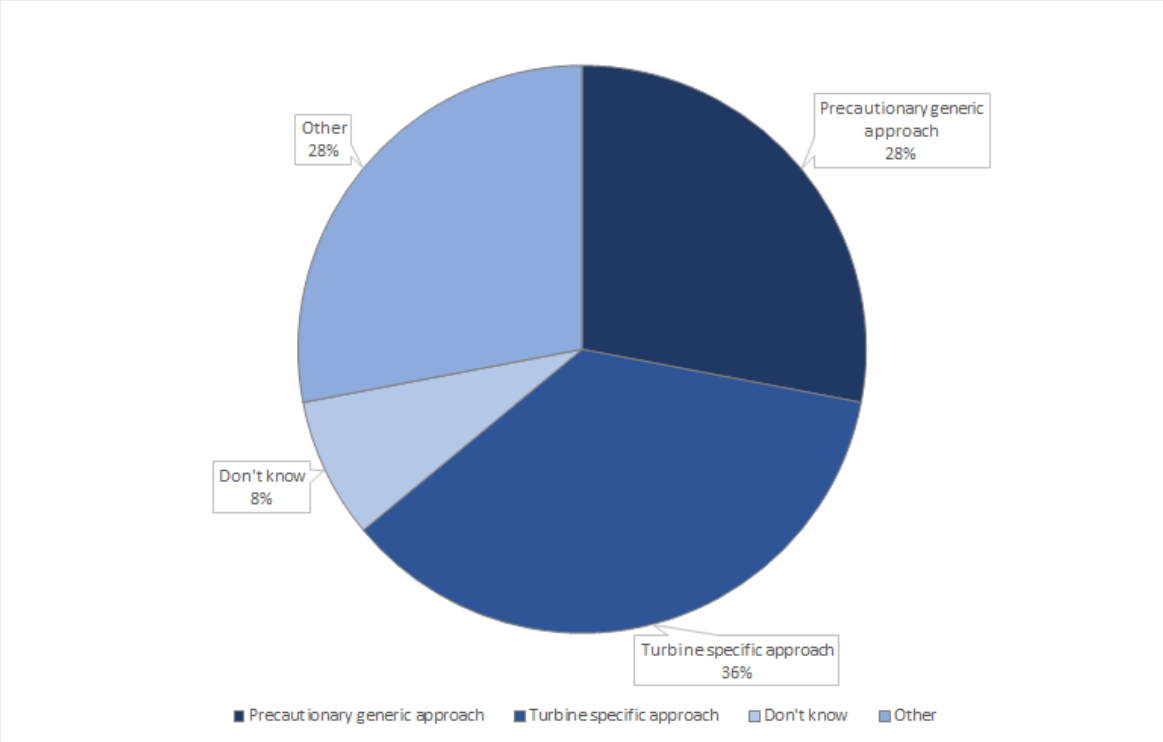


Figure 12 Relative proportion of stakeholder responses to question 8

74 Free text responses were very helpful, with most comments asking for both options to be available, even when stakeholders had selected either a precautionary generic approach or a turbine specific approach. Comments were also provided to highlight the issues around the commercial sensitivity of these data at a pre-construction stage, both from a developer's perspective, and a turbine manufacturer's perspective. Several comments received were about the need to provide these data and how these assessments should be undertaken, were beyond the scope of this project and were issues for regulators and their advisors to consider (e.g. Rochdale envelope approach to a generic or specific approach).

75 In this case, word cloud analysis (Figure 13) did not provide much useful additional value, as most of the commonly used words were from the question itself.

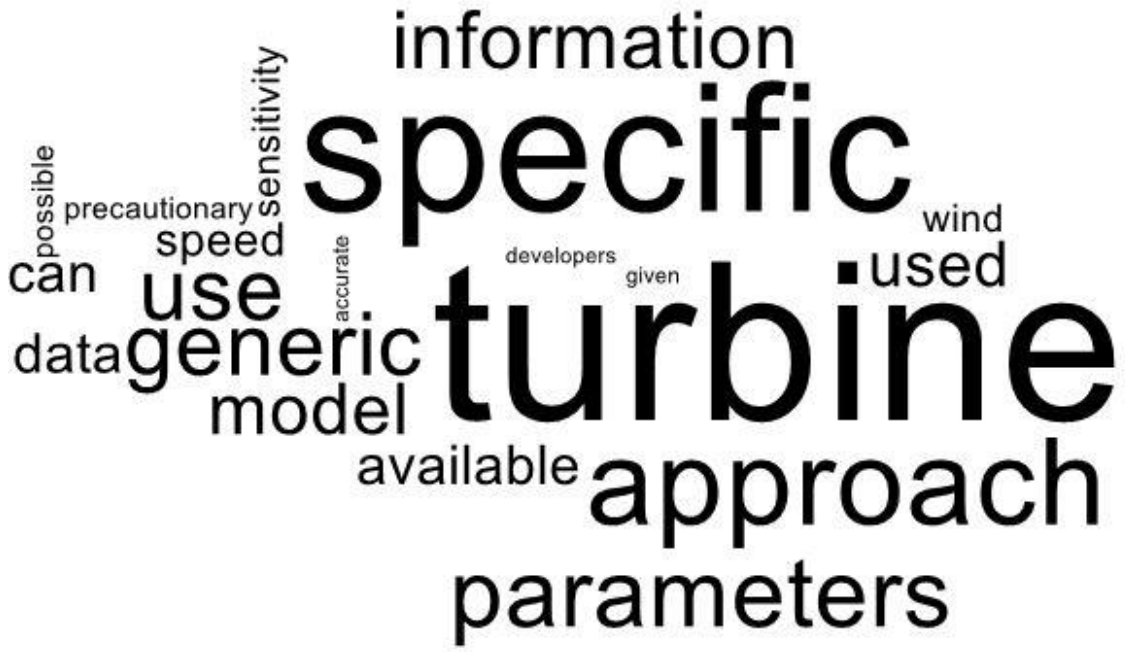


Figure 13 Word cloud of the free text component of question 8

76 Question 9 was also a single part question, "Do you think that the Band (2012) model (& Masden (2015) model) correctly calculates the probability of collision BEFORE avoidance rates are applied?"

77 It has been suggested, several times, in the past that the basic model calculations should be carefully checked by persons with a good understanding of mathematics. This may have been done, so it could be valuable to ask stakeholders this, in case someone has undertaken this check.

78 Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

79 A free text box asked stakeholders that responded "No" why they think that the model does not make the correct calculation.

80 Almost half (48%) of the responses were "No", that stakeholders did not think that the model made the correct calculation for the probability of collision (Figure 14). Only 16% responded that the model did make this calculation correctly, and more than a third (36%) did not know.

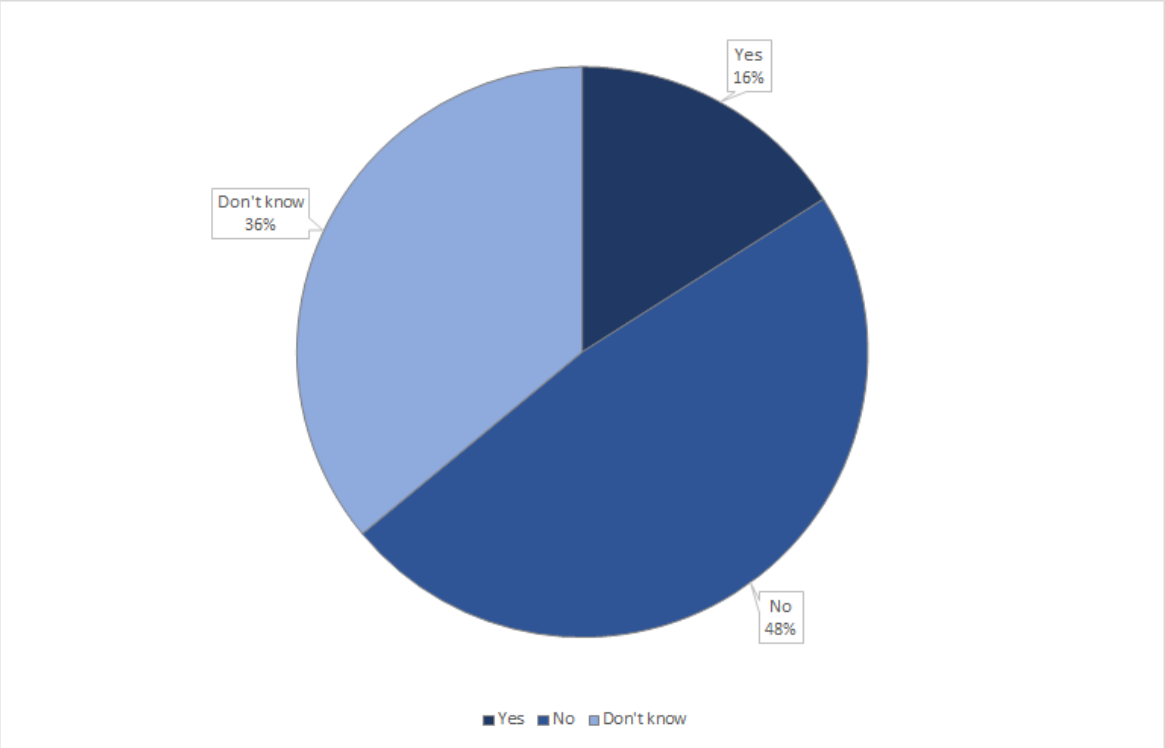


Figure 14 Relative proportion of stakeholder responses to question 9

81 Many of the free text responses commented that the calculation is a simplification and that as it is "just a model" it is by definition, likely to be wrong. Several other comments stated that the model was the best available, so within the assumptions made by the model it was making the correct calculations. Comments also included issues with the assumed 90° angle of approach, the lack of bird behaviour aspects and weather influences captured by the model. One comment suggested that the model flux calculation was likely to be incorrect as it's unbounded (in comparison to flow calculations). Overall, most comments, and the categorical responses, suggest that the question was inappropriately worded, as it was intended to draw out issues with the underlying mathematics, rather than other issues, such as available inputs.



Figure 15 Word cloud of the free text component of question 9

3.1.5 CRM outputs

82 Question 10 was the only question in the section on CRM outputs.

83 Question 10 was also a single part question, "Are there any outputs from the Masden (2015) model not currently provided that may be useful to include in a future model? (A description of the outputs is provided in paragraph 6 of the introduction)"

84 This was an open question to gather information on outputs that have not been considered to date. Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

85 A free text box asked stakeholders that responded "Yes" to provide the outputs that they considered useful.

86 Almost half of stakeholder responses were "Don't know" (48%), which is likely a reflection of the relatively small proportion of stakeholders who had used the Masden (2015) model. The remaining half of responses were approximately evenly split between "Yes" (24%) and "No" (28%) responses. All those that responded "Yes" provide some free text responses, and some "Don't know" responses also provide free text responses. The "Yes" responders requested improved outputs that include tabular data on probabilistic collision outputs (that are currently only provided as plotted data), improved box plot outputs (to include 95% confidence intervals), summarised input information and the predicted number of birds that do not collide in addition to the predicted number that do collide. The "Don't know" responses were limited to a request for probabilistic outputs rather than a single value (which the Masden (2015) model already does), and for sensitivity testing of the new stochastic CRM.

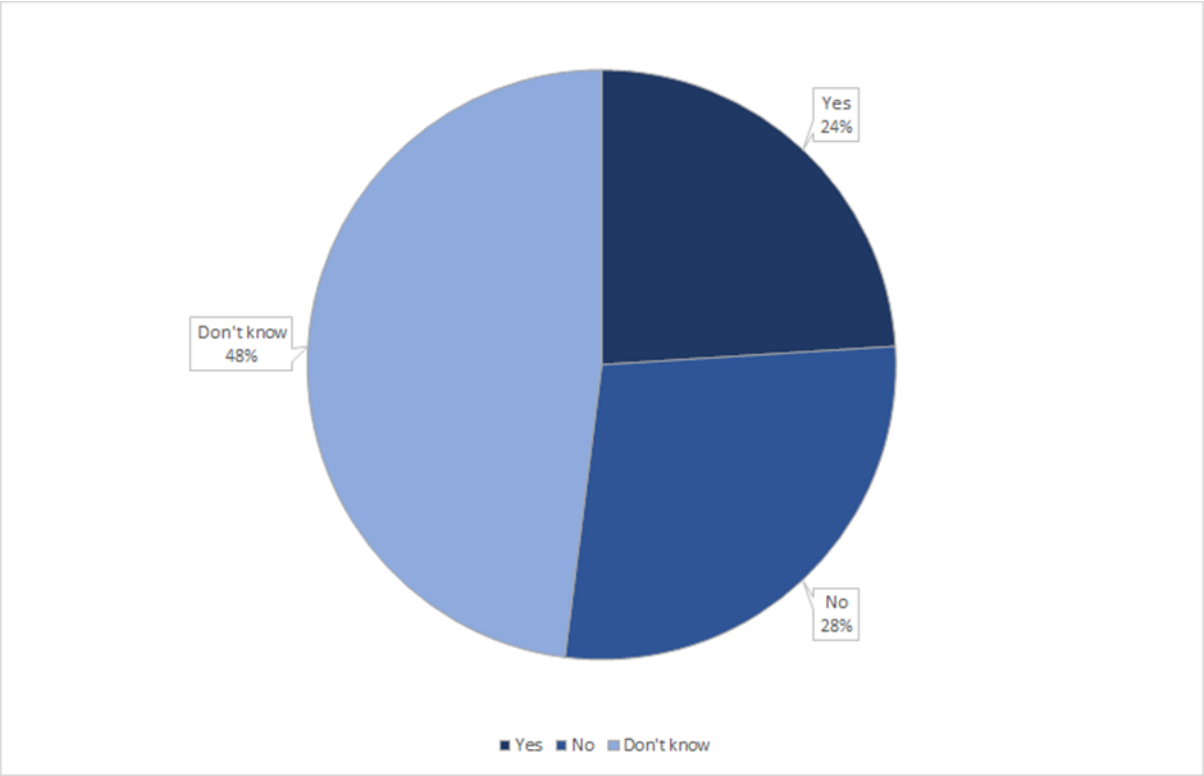


Figure 16 Relative proportion of stakeholder responses to question 10

87 The analysis of free text using a word cloud was not useful for Question 10, as response were too variable to find common themes.

3.1.6 CRM error checking

88 The only question in the section on error checking was Question 11.

89 Question 11 was also a single part question, "The current Band (2012) and Masden (2015) models do not provide any error checking. Is there any turbine specific error checking that would be useful to include in an updated Stochastic CRM?"

90 This question was particularly aimed at developers, hence the focus on turbine error reporting. It was agreed that there was sufficient ornithology expertise within the project steering group to provide advice on matters relating to the bird parameters in the model, but effectively no technical wind turbine experience. Stakeholders were provided with three possible responses:

- Yes;
- No; and,
- Don't know.

91 A free text box asked stakeholders that responded "Yes" to provide examples of useful error checking. Almost two thirds (60%) of stakeholders responded, "Don't know", which was likely a reflection of the nature of the question being turbine specific (Figure 17). About one quarter (28%) of respondents responded "Yes" and only 12% responded "No".

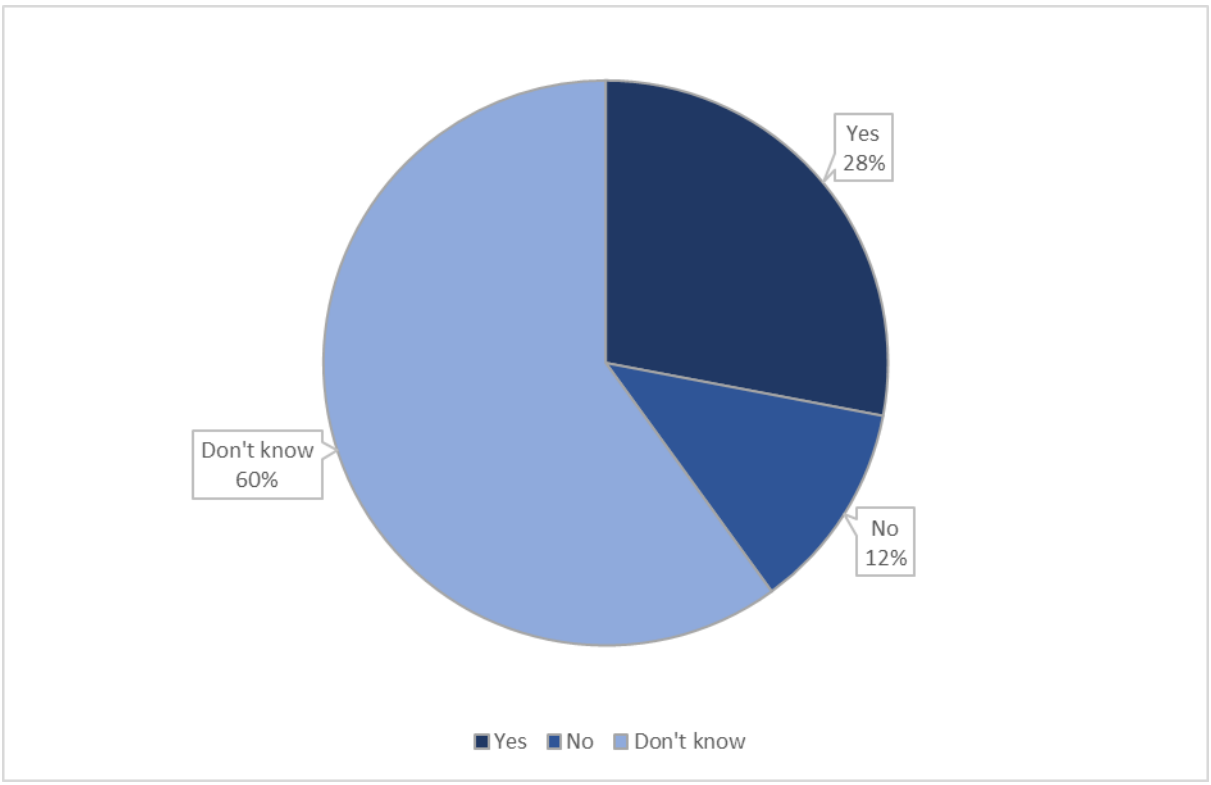


Figure 17 Relative proportion of stakeholder responses to question 11

92 Free text responses included requests for the model to flag up when parameters appear out of range, checking the numbers of birds estimated to pass through the rotor with the available population as a sense check and responses that indicated that the question did not provide an adequate explanation of its intended purpose to stakeholders'.

3.1.7 CRM improvements

93 There were two, free text only, questions in the section on CRM improvements, which were asking general questions and allowed stakeholders to provide any feedback they wished.

94 Question 12 was a single part question, with free text only, "What would be the main improvements you would like to see to a stochastic CRM? Please provide your order of preference/importance (highest first)." This question aimed to draw out practical changes that stakeholders think may be valuable from a new stochastic CRM.

95 There were eight areas where more than one stakeholder provided feedback on possible improvements. There were an additional eight areas where only one stakeholder provided feedback. The most common responses to question 12 were focused on model inputs. While many of these responses were regarding the need for better empirical data on model input values for birds (which was beyond the scope of this project), several were asking for the model to output a summary of the input values used in the model. There were also requests for default values to be provided in the model, but also that users should be able to change these.

96 The second most common comment to question 12, was for a user-friendly approach to modelling. It was clear from other responses that few stakeholders had much experience with using R, and a model

only being available in R could reduce the uptake of the approach among stakeholders. There were also a few appeals for the model to be available as R-code. The next most common response was related to model outputs. There were several recommendations for output summaries, as well as for outputs that provide the error around the estimate and also the probability distribution from the stochastic output.

97 The fourth most common set of recommendations from stakeholders were based around turbine information. Of all the comments provided on turbine inputs or outputs, only one was made by more than one stakeholder. This was in relation to the ability of the Masden (2015) model to use the relationship between wind speed, rotor speed and blade pitch of the turbines. If this element was to be retained in the model, stakeholders expressed a strong preference that default values should be used unless turbine specific parameters are publicly available.

98 A few comments were received about the lack of weather related effects on bird input parameters, though, since the purpose of this project is to create a working stochastic version of the Band (2012) model, this is not within the scope of this project. Similarly, there were a couple of comments regarding avoidance rate data that are used in the model, and this is also not within the scope of this project to address. There were requests for better flexibility in the application of seasonality within the model, though this is relatively easily addressed by users for the point estimates, as predicted collisions are additive, though errors are not.

99 Two comments were also provided regarding the slow speed running the Masden (2015) model, and requests for improved model running speed to be addressed. There were approximately eight different comments that were provided by single stakeholders, which varied greatly. These included comments about the calculations of flux of birds through the wind turbine, use of the oblique approach of birds to the turbine rotor, separate model runs for upwind and downwind flights (which can be done by users anyway) and for model validation.

100 Word cloud analysis (Figure 18) picked up on the multiple recommendations for stakeholders for better bird input values (beyond the scope of this project) and for the model to provide summaries of the model inputs. The requests for different model outputs were also reflected in the word cloud analysis. The word cloud did not pick up on the requests for a user-friendly version of the model, perhaps due to the way that stakeholders described this without using common terms. “Variation” was a relatively common word, which was related to both input values and to outputs. “Speed” was also found relatively frequently, which was related to both model speed, and bird flight speed as a user input.

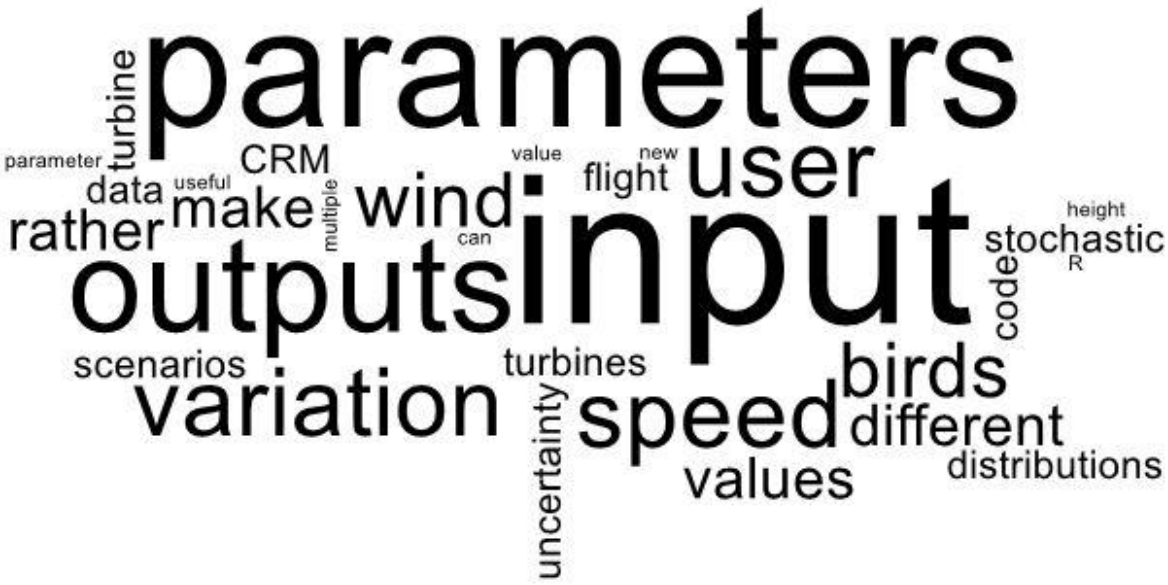


Figure 18 Word cloud of the free text question 12

101 Question 13 was a general free text response question, "Are there any other comments you would like to make about collision risk modelling?", designed as a catch all to ensure that stakeholders were able to provide any other feedback they wished.

102 Responses to question 13 were more variable than other questions, which was expected given the broad question asked. There were few comments made by more than one stakeholder. There were a few comments that the model should be transparent, and related to this a request that the R-code should be freely available. There were also several comments that the CRM is only a model, and there is often both too much precaution used in parameterising it, and too much faith placed in the results, that are often treated as more accurate a prediction than is likely to be true. Other useful comments included a request that single value outputs are no longer used and that only probabilistic outputs are considered, a recommendation is provided for the number of runs needed to produce a useful stochastic output, and that data from the ORJIP project could be used to sense check some of the model calculations. A request was made that care is taken to ensure terms are clear and consistent.

103 There were several other comments that, while useful, were out of scope for this project. These included more use of tracking data to inform bird input parameters, a better understanding of bird aerial density data and more consideration of the difference in weather conditions during surveys with the likely weather conditions when turbines are operational.

104 Word cloud analysis showed that there were many commonly used words (Figure 19), but due to the broad basis for the question there were no key messages that could be better elucidated from the word cloud.



Figure 19 Word cloud from the free text question 13.

3.1.8 Telephone interviews

The telephone interviews were intended to be short (30 – 60 mins) and allow for discussion and exchange of ideas in order to capture any useful additional information. There were four questions:

- Question 1: Did the questionnaire allow you to provide all the feedback you would wish to give? If not, what was missing and what feedback would you want to give?;
- Question 2: When the stochastic CRM is produced do you think you will use it? (If the interviewee is a developer or consultant, then ask: would the new sCRM need to be recommended by the relevant regulator and their SNCB for you to use it?);
- Question 3: Assuming the stochastic CRM is produced and works, what are the next new developments in CRM you would like to see? Are there any other comments you want to make about the survey or CRM for offshore wind farms?; and
- Question 4: How do you think you would implement the results from a stochastic CRM in to an impact assessment and a population model?

105 A total of eight interviews were conducted. Most were with environmental consultants (5), two with developers and one with an NGO. Overall the responses only underlined the comments made in the questionnaire itself.

106 In response to question 1, all of the stakeholders interviewed agreed that the survey was sufficient to allow all the feedback they wished to give. Several provided additional feedback at this stage, with the two most common comments relating to the slow speed of the Masden (2015) model, and the need to provide a user-friendly version as well as a coded version of the model. There were also comments on the value of the outputs including a tabulated summary of the inputs used.

107 Responses to question two all agreed that regulator, and SNCB, approval would be needed to use the model in consent application. However, several consultants noted that they would evaluate the model

anyway and would advise clients accordingly on the value, or otherwise, of the stochastic CRM. One stakeholder noted that the opinion of the RSPB on the model would also have some importance.

108 The most common responses to question three were the need to improve the empirical data on birds used as inputs, and the need to better incorporate information on bird behaviour in relation to weather. There were mixed messages from consultants and developers on the use of the relationship between wind speed, rotor speed and blade pitch. Some consultants noted that they had been provided with these data when asked, while developers noted that under Contracts For Difference ('CFD'), such information would not be readily shared in a public domain, highlighting the need for a generic approach.

109 Responses to question four were the most variable. Issues with the use of a mean and confidence interval around it were noted as problematic for regulators, and that guidance from SNCBs will be needed. One consultant noted that the existing models can give very precise outputs, that is far more than the accuracy of the model, so requested that outputs are always rounded up to the nearest whole bird (at least). Only one stakeholder requested tabular outputs of the collision probability from the model, to be used as an input to a stochastic population model. There were several comments about the CRM and population models being only model, so comparisons being of the most use.

110 Finally, the results of the telephone interviews, while not adding to any stakeholder requested changes to the CRM, did highlight the key messages from the survey.

3.2 Stakeholder requested changes

111 The results of the survey, while taking in to account the scope of the project, results in the following changes that have been requested by stakeholders:

- Create a user-friendly interface for non-R users;
- Speed up the code;
- The number of turbines should be a user input;
- Output predicted collision probability data;
- Provide summary of input values as an output;
- Seasonal (as well as monthly & annual) assessment (default + user defined);
- Error checking inputs and collision probability; and,
- Monthly or seasonal flight height inputs.

4 Comparison of Band (2012) and Masden (2015)

112 The Band CRM is implemented in two distributed forms: a deterministic version in Excel, based on macros and cell-to-cell calculations (Band 2012); and a version with stochastic elements, coded in R (R Core Team, 2016) by Masden (2015).

113 A comparison is presented here, based on general properties and on the outputs when both versions are run for the same scenario. The scenario considered was for a single species (gannet *Morus bassanus*) at a Scottish offshore location. The two implementations will be referred to as the Band and Masden implementations hereafter.

4.1 High level comparison

114 The interfaces to the two models are fundamentally different. The Band implementation is an Excel workbook, with all parameters and data presented cell-wise over numerous spreadsheets. There are effectively no checks on inputs (other than failure to compute), although some elements are protected from alteration. Being a spread-sheet, there is little in the way of an audit trail for presented outputs.

115 Interaction with the Masden implementation is via a main R script file, for high-level parameters, and a series of input files (comma-separated-value: CSV) for data and various parameter sets. Users require an installation of R, appropriate packages and some familiarity with running R code. There are effectively no checks on inputs other than failure to compute i.e. general warnings and errors from R.

116 The data/parameter requirements for the Masden implementation are larger, in keeping with its additional stochastic components e.g. bootstrapped flight heights, parameters governing statistical distributions on CRM parameters. The format of these files, such as column names, must be exactly as expected by the code, so templates need to be followed precisely.

117 Outputs from the Band implementation are tables and graphics within the Excel workbook. Outputs from the Masden implementation are files: CSV for tables and PNG graphics. The input data are also outputted from Masden, giving an audit trail for a particular set of outputs.

118 Calculations using the Band implementation are reasonably fast, on the order of a few seconds to run the imbedded macro for Option 3. However, the spreadsheet requires reconfiguring for each species and speculative turbine configurations. In contrast, the Masden calculations take substantive time. For example, a single species with 1000 Monte-Carlo iterations (a common modest number) might require an hour on a mid-range computer. This scales linearly with the number of species and turbine configurations e.g. two turbine configurations and 10 species might require almost a day of computer time. However, the species-turbine scenarios can be specified in advance, after which the program will iterate over all consecutively.

4.2 Output comparison

119 The principal output from both implementations is the predicted numbers of bird collisions – by month and a yearly total. These are presented for different avoidance rates, large-array corrections, species and "options" for the treatment of bird flight height distributions. The fundamental difference in outputs, is that Masden provides uncertainty in estimates. The uncertainty in collision risk is determined via Monte-Carlo (i.e. resampling of parameter values from statistical distributions on inputs) and expressed by standard deviations, coefficients of variation, inter-quartile ranges, box-plots, etc.

120 The calculations from Band and Masden implementations were compared by using identical inputs for common components and the suppression of Monte-Carlo variability, i.e. the stochastic Masden implementation was forced to provide deterministic predictions for comparability with the Band implementation. This allowed comparison of the basic calculations underpinning both.

121 Using Option 1 (the 'basic' Band model), the risk estimates for the Band and Masden models were deemed to be the same, within mild rounding errors. This indicated that the core functions for collision risk were providing effectively identical results.

122 In contrast, Options 2 & 3 (different treatments for flight height distributions) provided different results, with the Masden collisions estimates being somewhat higher and more consistent with Bands estimates with lower avoidance e.g. Masden's 95% avoidance estimates were similar to Band's 98% avoidance estimates.

123 The difference in results was mainly attributable to an apparent error in the Masden code, whereby the height of the turbine is incorrectly calculated when relating to the bird flight height distributions – effectively lifting the turbine higher. There may be further, more subtle, differences due to the bespoke visual basic 'interpolate' function found in Band, this being implemented differently in Masden.

4.3 Overview

124 Neither implementation is user-friendly, and both are prone to user errors. The current Masden code provides systematically different risk assessments for Option 2 & 3 calculations compared to the Band implementation – which is considered the standard here.

125 The Band implementation benefits from transparency of inputs, but a large, complex interface. There is little to check the validity of inputs, unintended alterations to the spreadsheet are opaque and there is effectively no audit-trail linking inputs to purported outputs.

126 In contrast, the Masden implementation might be considered more direct and efficient in user interaction, but requires interaction with R and is slow to calculate. There is similarly little to check the validity of inputs, but there is a reasonable audit trail linking the code run to the outputs presented. Failure of the code will produce esoteric R errors and would require modest R capabilities to resolve e.g. an error in the input parameter or data files.

5 Coding a new stochastic CRM

5.1 Code review

127 The Masden code was subject to a line-by-line evaluation. Broadly the following was found:

- There is a lack of consistency of coding, suggesting multiple authors, given markedly non-standard approaches.
- The code is inefficient, relying on multiple nested loops for its calculations, rather than vectorised approaches. Related to this, there is a repetition of objects which creates confusion due to synonyms.
- Scoping is poorly conceived in places, where functions rely heavily on global objects.

128 The code benefitted from substantial re-writing for efficiency, consistency and clarity.

5.2 Recoding

129 The Masden code was recoded, with the main goals of improving usability (including speed), transparency and robustness – as well as bug fixes and alterations in light of recent reviews of the code (Trinder 2017 and our detailed code review). These were achieved by creating a user-friendly Graphical User Interface (GUI) to interact with the code and progressively streamlining and improving the structure of the underlying code.

130 The code was moved to a version control system (GIT) and improved in stages. This provides a detailed audit-trail of modifications and reversion to any state is possible. Other developers can collaborate or take over future development relatively seamlessly.

131 There has been vectorisation of many elements to improve speed and readability. Coding consistency has been improved and redundant objects removed. Revised distribution options have been provided for the Monte-Carlo to address the points raised in Trinder 2017.

132 Default parameter values are provided and the inputs are either constrained or flagged to the user if unreasonable. Data can be provided directly through the GUI or from the uploading of template data files. Pop-up help text is provided throughout along with guidance for use.

133 The GUI has been developed in Shiny, a set of R tools that create HTML interfaces to R code. This has many benefits:

- It provides a user-friendly GUI that users access through a standard web-browser – all R code is invisible and no direct code interaction is required;
- It is free and open-source, there is no vendor lock-in;
- The underlying R code is maintained on a remote server that all users connect to. Any alterations are immediately realised for all users. No installation or maintenance of R is required by users; and
- There is a wide-range of ways that input and output can be specified, to suit users.

5.3 GUI implementation

134 General information about Shiny can be found on https://shiny.rstudio.com/. The current version of the GUI can be found at https://dmpstats.shinyapps.io/avian_stochcrm/ and the following gives a brief indication of its use.

135 The workflow is broken into four main steps. In the first instance we set turbine parameters for the wind-farm. The GUI provides sliders and fields for all parameters and plots the implied parameter distribution in each case (Figure 20). Default values are presented and where appropriate field values are constrained e.g. counts are non-negative. In addition, ranges of plausible parameter values were solicited from the Project Steering Group (PSG). Entry of values that are not impossible, but outside expected ranges may elicit warning messages.

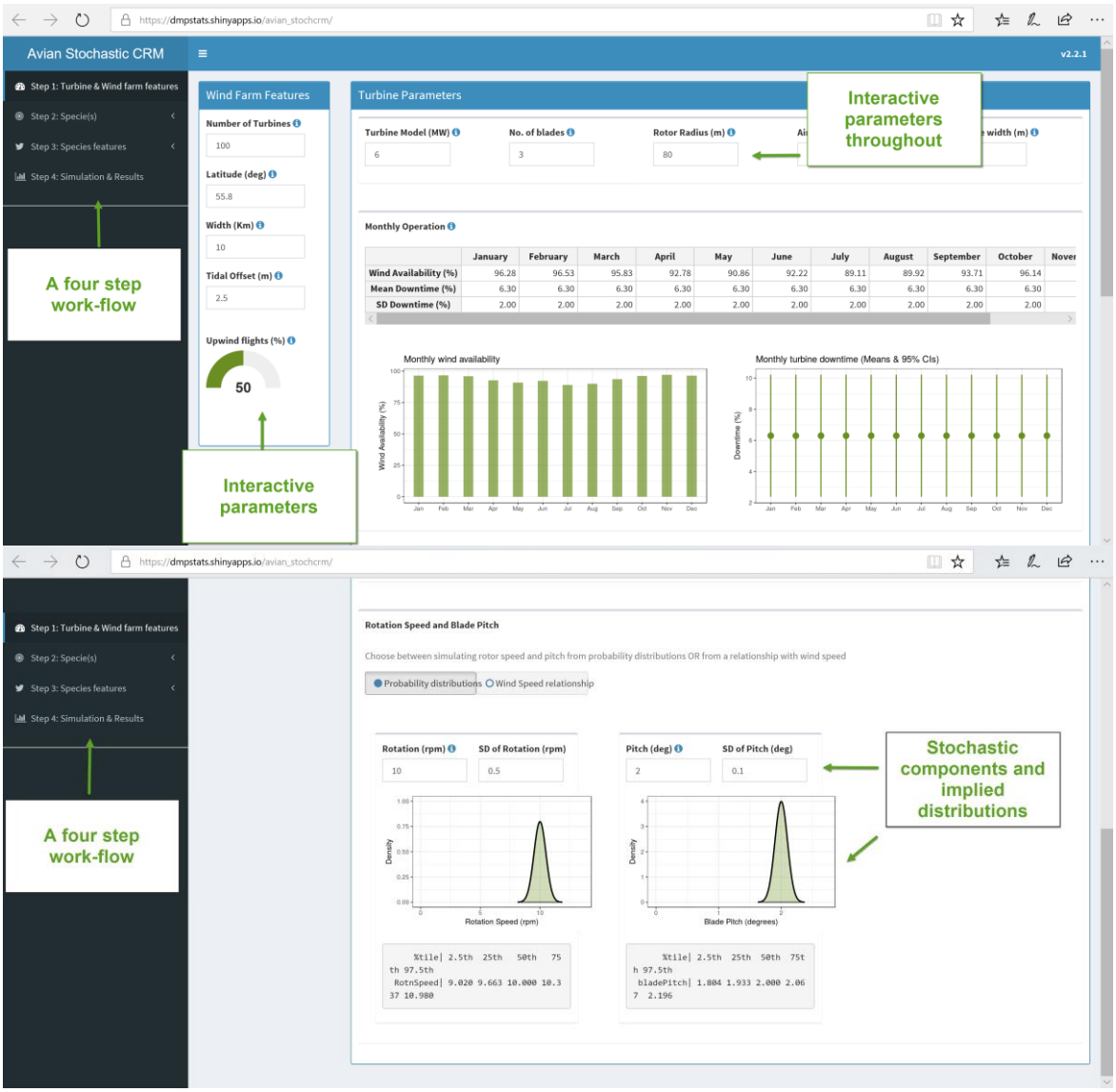


Figure 20 The GUI introduction page. Turbine parameters

136 Additional options have been added for flexibility in portraying relationships between wind-speed and the turbine's rotor pitch and speed.

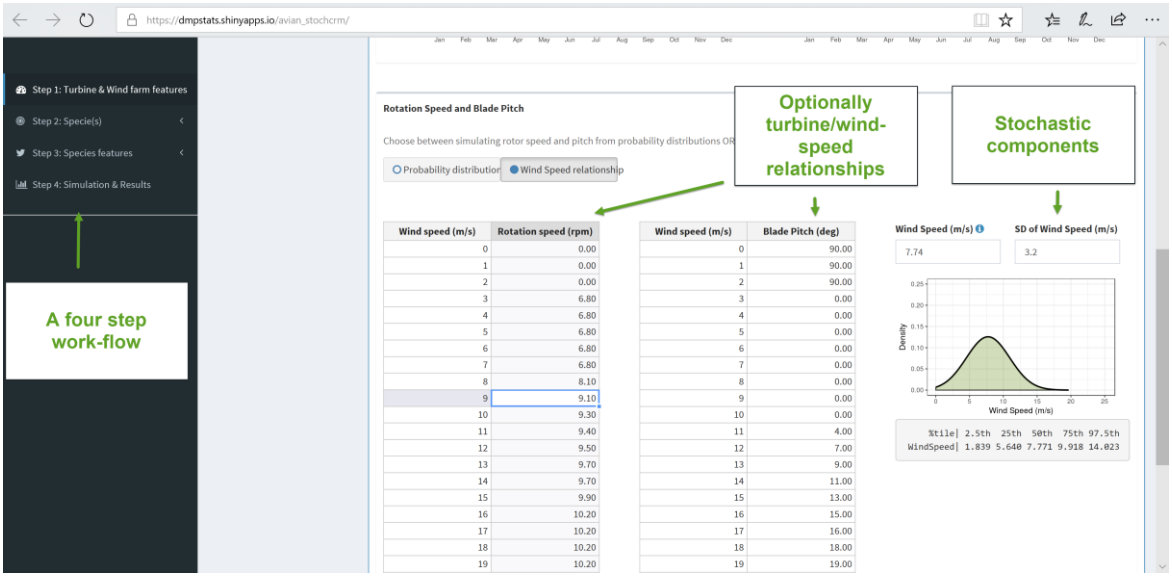


Figure 21 The GUI introduction page. Turbine parameters

137 After setting the turbine parameters, noting there may be several proposed turbine setups (Figure 22), the species of interest are selected. Currently these are pre-defined, as there are limited datasets stored for the flight-height distributions, as described in Masden (2015). Further species can be added if equivalent data is available.

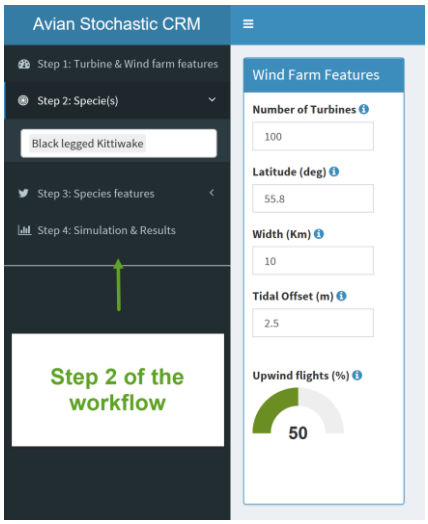


Figure 22 There are four basic steps – defining turbine parameters, species to consider, species parameters, then the size of simulation, before results.

138 Each of the species have parameter sets that are defined (Figure 23). As before, parameters for the stochastic components are set and the implied distributions are plotted. Entries are constrained to avoid impossible values and offers warnings if entered values are outside expectations, as per the PSG opinions.

139 A number of modifications have been made with respect to bird densities and their stochastic treatment, in line with the findings of the review by Trinder (2017). The previous default treatment by truncated Normal is retained, but with the upper truncation value removed. Further, users may offer an estimate and confidence bounds or a general series of reference points for whatever distribution they think applies.

140 Bird flight height distributions similarly have a range of options: a single flight height distribution as previously held in the Masden code, or one of the user's choosing; alternatively, bootstrap flight height distributions as previously held in the Masden code, or a set of the user's choosing. Templates can be downloaded from the app to ensure conformity of input data when uploaded.

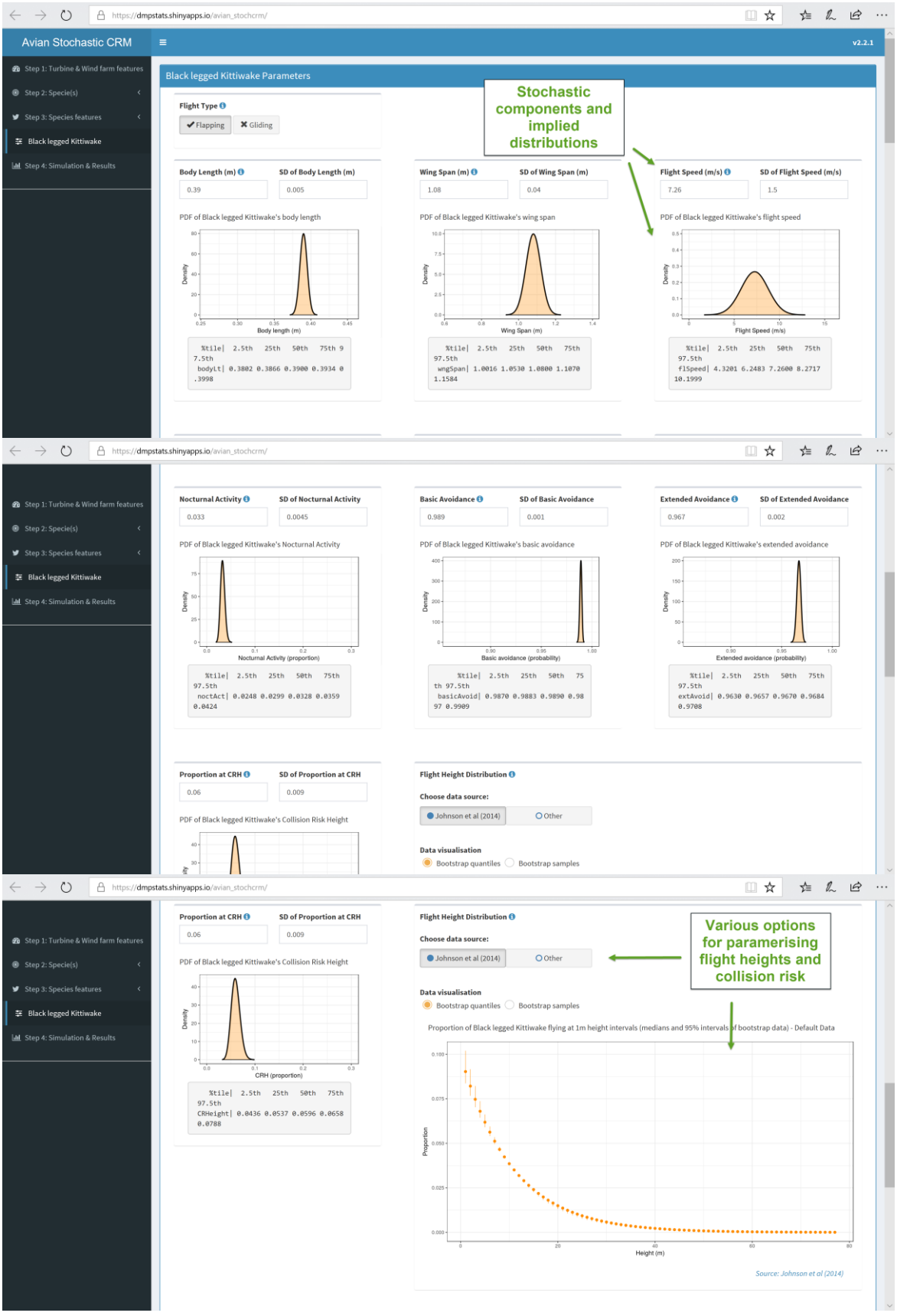


Figure 23 The interface is similar throughout – interactive parameter setting then a graphic showing what is implied.

141 The simulation is set in motion – the amount of time required being proportional to the number of turbines, species and simulation iterations (Figure 24).

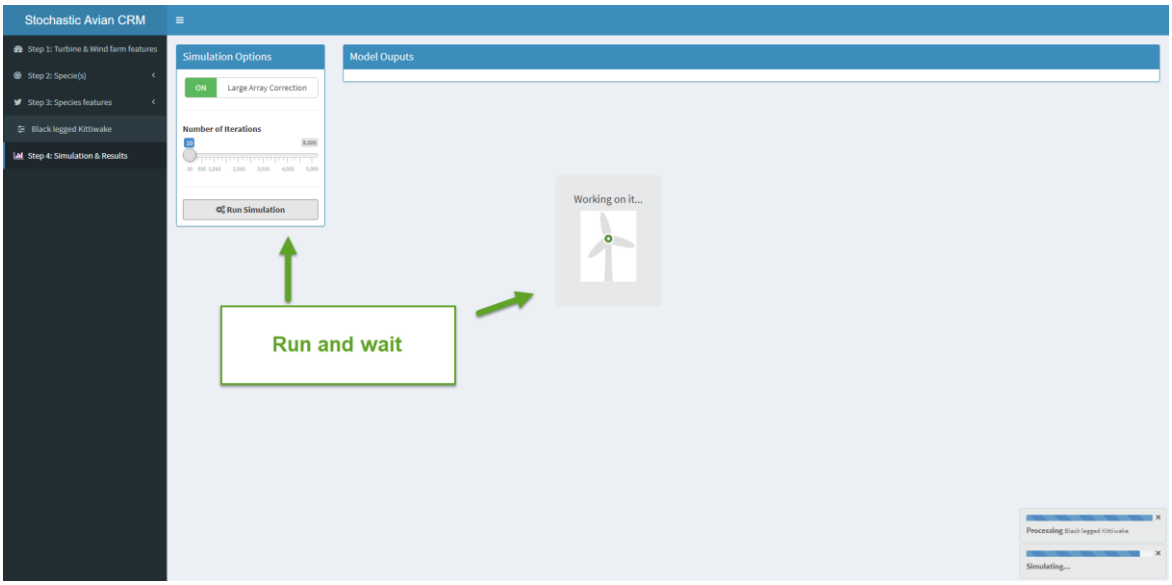


Figure 24 The final step is setting the number of iterations and large-scale corrections.

142 Outputs are extensions of those of Masden, albeit rendered in HTML and available as downloads (Figure 25).

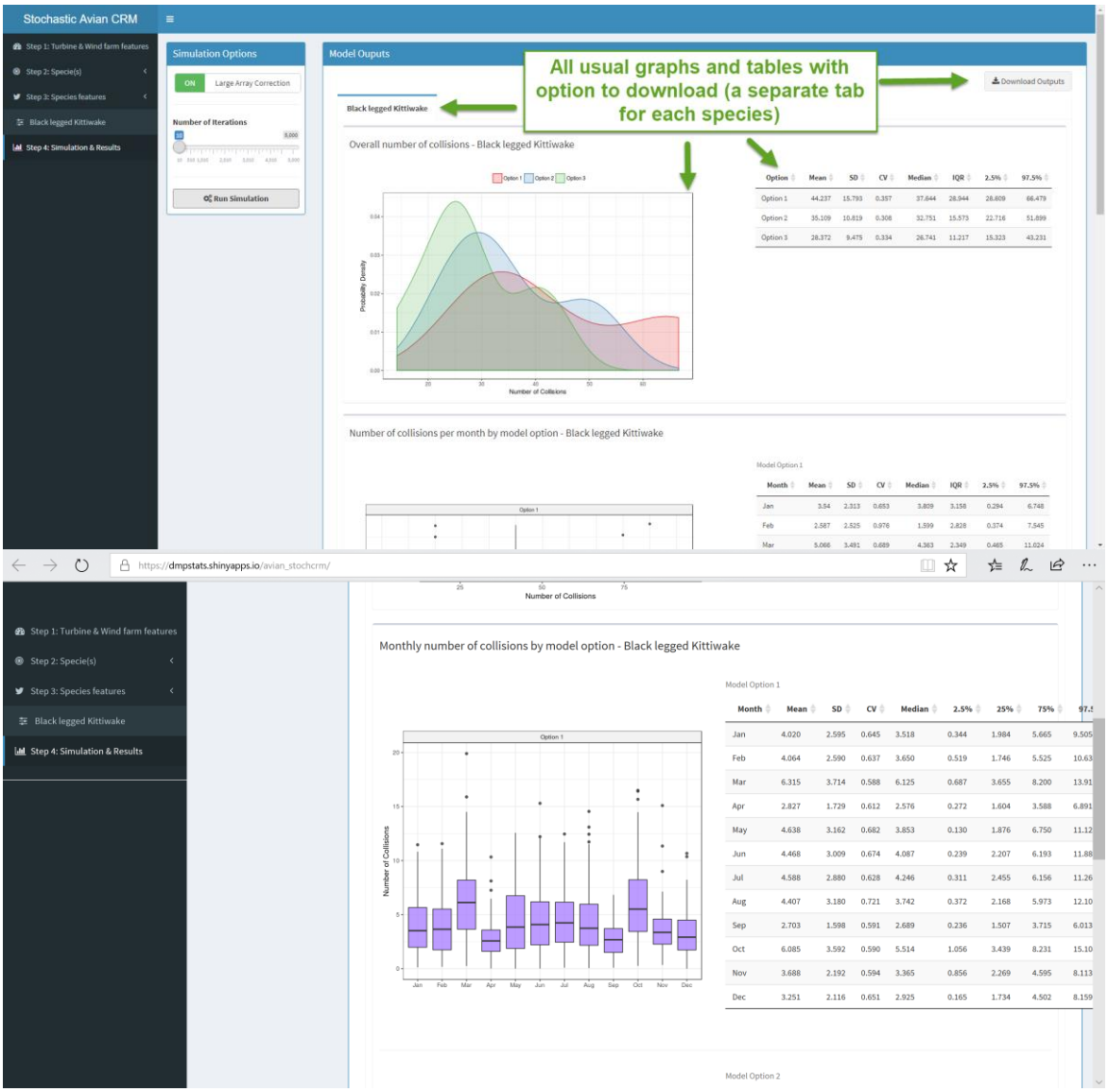


Figure 25 The results are tables and plots similar to those in Masden, rendered in the GUI. There are download options.

6 Testing of new stochastic CRM code

143 The new code was tested by its conformity with Masden 2015 outputs and the point estimates of Band 2012. Where disagreement was found between Masden and Band, the Band results were assumed correct and the new CRM code conforms to this.

144 The GUI was further tested by the presentation of extreme and corrupt inputs (including data-files) to ensure sensible behaviour.

7 Conclusions

146 To address the identified need for improved modelling of stochastic variation in collision risk modelling of seabirds for offshore wind farm development applications a stakeholder survey was used to inform the changes needed to create a new stochastic CRM.

147 The stakeholder survey identified seven key changes needed to the currently available CRMs. These included a user-friendly interface, full data outputs, seasonal inputs and assessments, error checking and flexibility for users to change default values.

148 These changes were implemented by experienced R-code developers through the updating and streamlining of the existing Masden (2015) code. The key changes requested by stakeholders were implemented, along with the recommendations of Trinder (2017).

149 A user-friendly interface was developed by coding these models into a Shiny app in R (app version 2.2.1 at time of reporting found at https://dmpstats.shinyapps.io/avian_stochcrm/) that allowed users to easily input values for turbines and birds and incorporated default values and guidance to reduce human error. Flexibility was maintained by allowing users to use non-default values.

150 There are two variants of the revised stochastic CRM, both coded in R. Both provide the full GUI interface via shiny as outlined. The online version runs on the Shiny server, while a downloadable version will run locally on the computer it is installed on, using the internet browser on that computer. It can be downloaded from https://github.com/dmpstats/stochCRM.

151 Data outputs from the Shiny are provided both graphically and as a data download. This provides end users with all the information needed to interpret the collision risk values, and their uncertainty.

152 Both the R-code variants of the sCRM are a highly flexible, stochastic model that provides a prediction of seabird collisions with a correctly calculated error estimate for use in Environmental Impact Assessments.

8 Contributions and Acknowledgements

The project was managed by Dr Ross McGregor (HiDef Aerial Surveying), who also produced and analysed the questionnaire with Sue King (Sue King Consulting). The CRM shiny app and R code were produced by Dr Carl Donovan (DMP Statistical Solutions) and Dr Bruno Caneco (DMP Statistical Solutions). The project was overseen by Andy Webb (HiDef Aerial Surveying).

The authors would like to thank all of those stakeholders who provided a response to the questionnaire, and to those who also agreed to follow up telephone interview. The high response rate to the requests for feedback was greatly appreciated.

The invaluable input to the entire project process from the Project Steering Group greatly aided in the success of this project. The Project Steering Group comprised: Jared Wilson (Marine Scotland Science), Helen Wade (SNH), Julie Black (JNCC), Aly McCluskie (RSPB) and Mel Kershaw (Natural England).

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Appendix I Questionnaire *Pro Forma*

An improved stochastic Collision Risk Model for seabirds in flight

Stakeholder Questionnaire

1 OF 11

1 Introduction

1 Collision Risk Models (CRM) are used to predict the number of birds that may collide with wind turbines. This project will develop an improved stochastic CRM for seabirds by consulting with stakeholders to determine their requirements and then develop CRM software to meet those needs.

2 Phase one of this project is to identify existing gaps in the CRM process through stakeholder engagement. This questionnaire asks 13 questions in 8 sections covering each part of the CRM process. There are free text boxes at the end of the questionnaire, where you can provide feedback on which improvements you think would be most important and any other feedback you may wish to give to the project delivery team. Please share this questionnaire with any other individuals in your organisation that may wish to respond. We would prefer one response per individual, rather than per organisation.

3 The overall aim of this project is to produce a user-friendly CRM that uses the variability in the input values to produce an output that gives a range of output values around some central point (mean, median). The CRM will be coded in 'R', providing a great deal of flexibility to users. However, this is not friendly to all users, so a graphic interface (https://shiny.rstudio.com/) could be produced if stakeholders considered it useful. The CRM would be tested against the original Band (2012) model, to ensure that it provided the same calculations, but with the added advantage of variability in input values being able to produce an output with suitable and useable variability.

4 This questionnaire does not ask any questions about the bird input parameters. It is intended that the improved stochastic CRM will provide default values based on the species selected, but these will also be user definable.

5 The questionnaire specifically asks for feedback on turbine input values and the usefulness of these. Many of the input parameters may require both a mean value and the standard deviation around that mean. Feedback is sought on both the input values themselves and the standard deviations around them. The input values required from the Masden (2014) stochastic CRM are:

- Number of turbine blades;
- Turbine rotation speed (r.p.m.) – this is the speed the turbine will operate at most often;
- Turbine rotor radius – measured from the axis of rotation to blade tip, not just the blade length;
- Turbine hub height – Sum of the rotor radius and minimum blade clearance above Highest Astronomical Tide (HAT);
- Maximum blade chord width;
- Blade pitch – in degrees relative to rotor plane; and,

- Turbine operation time – this needs to be provided by month, and is the combination of predicted time spent operation from the predicted wind resource, and the planned and emergency maintenance down time.

In addition, the Masden (2014) stochastic CRM requires inputs on the relationship between wind speed (ms^{-1}), rotor speed (rpm) and blade pitch (degrees).

6 The Masden (2014) CRM also provides tabular and figure outputs. These are:

TABLES

1. Overall summary table of collisions by species, turbine and model option. Results are presented as monthly mean, standard deviation (SD) and coefficient of variation (CV), and median and inter quartile range (IQR).
2. Monthly summaries of collisions. Separate tables are produced according to species, turbine and model option for example 6_Black_legged_Kittiwake_monthlySummaryOpt3.csv. Results are presented as mean, standard deviation (SD) and coefficient of variation (CV), and median and inter quartile range (IQR).
3. Summary of sampled bird parameters by species, turbine and model option presented as mean and standard deviation (SD), and median and inter quartile range (IQR).
4. Summary of sampled turbine parameters by species, turbine and model option presented as mean and standard deviation (SD), and median and inter quartile range (IQR).

FIGURES

1. 3-panel boxplots of monthly collisions for model options 1, 2 and 3 by species, and turbine type.
2. Density plots of numbers of collisions by species, and turbine type. A density curve is plotted for each of the 3 model options.
3. If 2 or more turbine models are included, then a 3-panel figure will be produced for each species, with the panels representing model options 1, 2 and 3 and each panel containing density plots for the different turbines included.

7 To ensure that we fully capture the existing issues stakeholders have with the existing CRMs (i.e. Band 2012 and Masden 2014) a selection of stakeholders will be invited to participate in a short telephone interview.

8 Once completed the survey should be returned to ross.mcgregor@hidefsurveying.co.uk

2 Questionnaire

2.1 Personal information

Name	
Organisation	

2.2 CRM Concept

Question 1. Do you think that CRM is a useful method for assessing potential impacts from offshore wind farms?		
Yes	☐	If Yes, please describe the benefits of CRM. If No, please describe why you think that CRM is not a useful method.
No	☐	
Don't know	☐	

2.3 Experience

Question 2. How would you describe your experience in using the Band (2012) CRM for offshore wind farms? (Please select *all* that apply to your current *and* past use of the Band (2012) CRM).			
Model user Please tick if your primary role is using the Band (2012) CRM.	☐	Expert Experienced using the Band (2012) CRM in Excel in relation to consent applications	☐
		Occasional Have occasionally used the Band (2012) CRM for technical reporting, or undertaken QA of work done by others	☐
		Basic Have used the Band (2012) CRM once or twice, but never in relation to technical reporting or consent applications	☐
		None Have never used the Band (2012) CRM	☐
Model output interpretation Please tick if your primary role is interpreting the output from the Band (2012) CRM.	☐	Supervisory Supervise others in the use of Band (2012) CRM.	☐
		Reviewer Reviewed others use of Band (2012) CRM.	☐
		Other Please describe your use of the Band (2012) CRM in the box below	☐
		If other, please describe your use of the Band (2012) CRM)	

Question 3. What level of R user do you consider yourself to be?		
Expert	☐	If other, please describe your experience with R
Regular	☐	
Occasional	☐	
Never	☐	
Other	☐	

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Question 4. Have you ever used the Masden (2014) stochastic CRM (or another stochastic CRM) in R?		
Yes (Masden (2014) CRM) (if yes, please go to Q.5)	☐	If you have used another sCRM, or selected other, please describe your use of a stochastic CRM here.
Yes (Other stochastic CRM)	☐	
No (if no please go to Q.6)	☐	
Other	☐	

Question 5. Have you ever experienced issues running the Masden (2014) stochastic CRM (or another stochastic CRM) in R?		
Yes.	☐	If yes, what problems have you experienced and can you suggest any solutions to these problems?
No	☐	
Don't know	☐	

2.4 CRM Inputs

Bird parameters

Question 6. Are there any input values for birds (e.g. wind span, length, flight speed, nocturnal activity) that you think should be changed, improved or added?		
Yes.	☐	If yes, please describe which values should be changed or improved and why.
No	☐	
Don't know	☐	

Question 7. Should the new stochastic CRM retain all of the model Options (1, 2, 3 & 4) described by Band (2012)?		
Yes	☐	In no, please describe why and what changes you would like to see to the modelling of the different options.
No	☐	
Don't know	☐	

Turbine parameters

Question 8. The Masden (2014) CRM includes the relationship between wind speed, rotor speed and blade pitch. Given the commercial sensitivity of this information, should a precautionary generic approach be used or should turbine specific data used for consent applications?		
Precautionary generic approach.	☐	Please explain why you gave this response.
Turbine specific approach	☐	
Other	☐	
Don't know	☐	

2.5 CRM Operation

Question 9. Do you think that the Band (2012) model (& Masden (2014) model) correctly calculates the probability of collision BEFORE avoidance rates are applied?		
Yes	☐	If no, please describe why you think that the current model does not make the correct calculation.
No	☐	
Don't know	☐	

2.6 CRM Outputs

Question 10. Are there any outputs from the Masden (2014) model not currently provided that may be useful to include in a future model? (A description of the outputs is provided in paragraph 6 of the introduction).		
Yes	☐	If yes, please describe the additional outputs that would be useful.
No	☐	
Don't know	☐	

2.7 CRM error checking

Question 11. The current Band (2012) and Masden (2014) models do not provide any error checking. Is there any turbine specific error checking that would be useful to include in an updated Stochastic CRM?		
Yes	☐	If yes, please provide any examples of useful error checking.
No	☐	
Don't know	☐	

2.8 CRM improvements

Question 12. What would be the main improvements you would like to see to a stochastic CRM? Please provide your order of preference/importance (highest first).

Question 13. Are there any other comments you would like to make about collision risk modelling?

THANK YOU FOR COMPLETING THE SURVEY.

3 References

Band, W. 2012. Using a collision risk model to assess bird collision risks for offshore windfarms. Report to The Crown Estate (Strategic Ornithological Support Services); Project SOSS-02. British Trust for Ornithology, Thetford, UK.

Masden, E. 2014. Developing an avian collision risk model to incorporate variability and uncertainty. NERC.

Appendix II Telephone survey *Pro Forma*

MS Stochastic CRM stakeholder survey - follow up interview questions

Name	**Organisation**
Date	**Time**
Form of interview (e.g. phone, in person, via email, etc.	
Ask the interviewee if they have completed the survey questionnaire. **Yes** ☐ **No** ☐ If the interviewee has not completed the survey, walk them through the questions and record the answers for them.	

Question 1: Did the questionnaire allow you to provide all the feedback you would wish to give? If not, what was missing and what feedback would you want to give?

Question 2: When the stochastic CRM is produced do you think you will use it? (If the interviewee is a developer or consultant, then ask: would the new sCRM need to be recommended by the relevant regulator and their SNCB for you to use it?)

Question 3: Assuming the stochastic CRM is produced and works, what are the next new developments in CRM you would like to see? Are there any other comments you want to make about the survey or CRM for offshore wind farm?

Question 4: How do you think you would implement the results from a stochastic CRM in to an impact assessment and a population model?

Thank the interviewee for their time.