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Subject: Hornsea Project Three (UK) Ltd response to Deadline 6 (Part 4)

Dear Kay, K-J

Please find attached the 4th instalment of documents.

Best regards,
Dr Dominika Chalder PIEMA
Environment and Consent Manager

[redacted]

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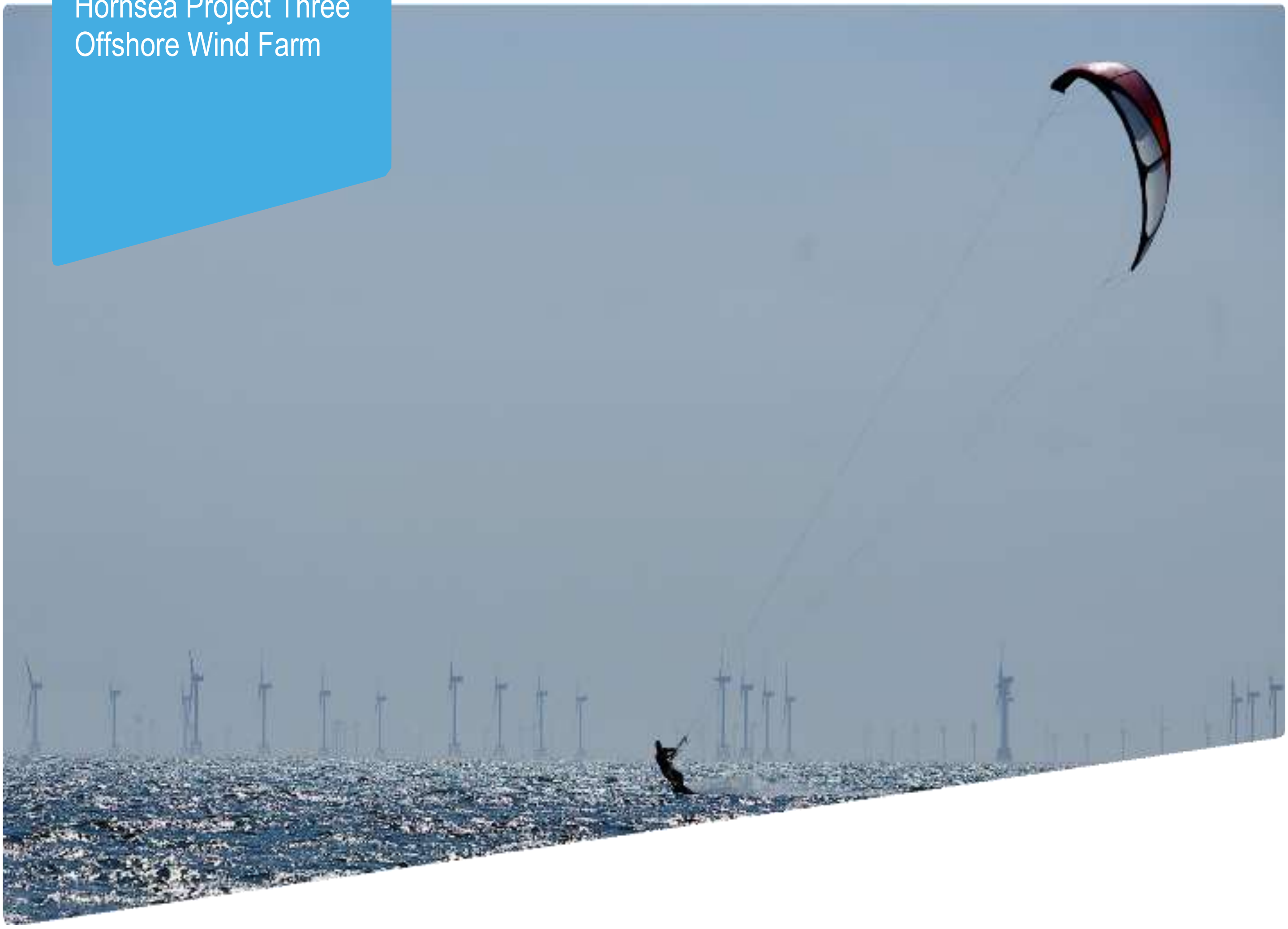
Orsted

Hornsea Project Three
Offshore Wind Farm



Hornsea Project Three

Offshore Wind Farm

Appendix 9 to Deadline 6 submission – Trinder M., 2017

Date: 8th February 2019

Hornsea 3
Offshore Wind Farm

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Document Control			
Document Properties			
Organisation	Ørsted Hornsea Project Three		
Author	Trinder M., 2017		
Checked by	n/a		
Approved by	n/a		
Title	Appendix 9 to Deadline 6 submission – Trinder M., 2017		
PINS Document Number	n/a		
Version History			
Date	Version	Status	Description / Changes
08/02/2019	A	Final	Submitted at Deadline 6 (8th Feb 2019)

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MacArthur Green

THE CROWN ESTATE

Estimates of Ornithological Headroom in Offshore Wind Farm Collision Mortality

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Document Quality Record

Revision	Date	Originator	Checker	Approved	Status
Draft	15/12/2016	Mark Trinder	Bob Furness		Draft
Minor revisions and additions	13/01/2017	Mark Trinder	Jess Campbell		Draft
Final edits and additions	24/02/2017	Mark Trinder			Final

Published by The Crown Estate.

Executive Summary

Assessment of the potential impacts of offshore wind farms include predictions of seabird collision mortality, for both individual projects and also cumulatively with other wind farms. Wind farm planning applications use worst case parameter estimates in collision risk modelling to ensure consented project designs are robust to subsequent modifications. For collision risk assessment this typically equates to designs with the highest rotor swept area for the planned generating capacity (i.e. many, small turbines). Currently there is no mechanism by which a wind farm's collision mortality can be updated to reflect the design changes. Consequently, these published collision estimates are the ones used by later wind farms in their cumulative assessments, even though the built wind farms may in fact present much lower risks of collision. As the number of wind farms increases there is therefore a growing risk of planning refusal on the grounds of unacceptably high cumulative collision risk.

To assist The Crown Estate to understand how much potential wind capacity is 'locked up' in the current cumulative totals MacArthur Green was commissioned to calculate updated collision mortality which reflected actual wind farm designs and thereby determine the ornithological 'headroom' – the difference between the two estimates. The methods used are detailed in a previous report (MacArthur Green 2016).

This work has focussed on five key collision risk species (gannet, kittiwake, lesser black-backed gull, great black-backed gull and herring gull), and one less widespread but potentially sensitive one (Sandwich tern). On the basis of current data, it is estimated that updating wind farm data would reduce cumulative gannet mortality by around 14%, while that for lesser black-backed gull would be reduced by around 40% (with the other species falling in between these two). On the basis of collision mortality per MW of installed (or planned) capacity, it appears that revising collision estimates could free up around 2,000MW of new wind farm potential in the North Sea. The data and calculations are provided in a spreadsheet which can be updated as new information becomes available. These should be treated as indicative estimates of headroom and potential new capacity, based on current methods and data availability. Certainty on wind farm designs and methodological changes will modify these figures in the future.

Reassessment has also been conducted for Special Protection Area breeding populations, in order to provide guidance on areas of higher and lower sensitivity for future development. It is important to stress that these can only provide a relative guide, and that future developments will still need to undertake full assessments, the results of which cannot be predicted.

It is also important to note that the acceptability (or otherwise) of cumulative mortality estimates is dependent on the status of the populations in question. Gannet populations are increasing and unlikely to be of conservation concern in the near future. In contrast, kittiwake populations have been in decline for more than a decade and this species can therefore be expected to be of increasing conservation concern both generally and with respect to wind farm impacts. Further declines may offset any headroom gained through the reassessment undertaken here.

CONTENTS

1. Background

As the manager of UK rights for offshore wind generation, The Crown Estate plays a major role in the offshore wind energy industry. As a responsible estate manager, The Crown Estate is interested to understand how much potential wind farm capacity is currently 'locked-up' in existing wind farm consents. This results from differences between impact assessments for proposed wind farm designs, which are typically derived using worst-case options for turbine dimensions and numbers, and as-built wind farms, which to date have invariably been smaller or make use of advancements in turbine technology to achieve planned power generation with fewer, larger turbines.

For any given wind farm the reduction in predicted collision mortality due to design changes may be quite modest. However, when these are summed across wind farms as is required for cumulative impact assessment (CIA), the reduction in mortality can become substantial. Nevertheless, attempts to apply these changes in CIA for wind farms have been deemed inadmissible because while the planning consent remains valid there is the potential that further wind farm development could be undertaken. The consequence of this is that ornithological CIA is based on the mortality predicted for the consented wind farm development.

This report presents an overview of work undertaken to collate turbine parameters for consented and built offshore wind farms in order to calculate the difference in collision mortality between the two. This difference is hereafter referred to as headroom.

As well as an overall estimate of headroom, this has also been considered at a finer spatial resolution, since the focus of ornithology impact assessment often falls on seabird breeding colonies which have been designated as Special Protection Areas (SPAs). Thus, headroom has been calculated at the scale of, for example, the North Sea and also within the foraging ranges of species of seabird which are considered to be at significant collision risk from key SPAs.

The results presented in this report and the accompanying spreadsheet are intended for internal use at The Crown Estate, in order to allow a better understanding of ornithological headroom which may be available across the existing offshore wind portfolio. The distribution of this report and its content for information is at the discretion of The Crown Estate.

2. Introduction

Ornithological assessment for offshore wind farms typically focuses on the potential for mortality as a result of collisions. Wind farms are assessed on the basis of both their project alone impacts and also cumulatively with other wind farms (and other relevant developments) with which their effects may be combined. As the number of offshore wind farms increases so do the cumulative impacts and as a consequence it seems reasonable to conclude that, at some future point, wind farm applications will be rejected on the basis that no more mortality is permissible. While this implies that statutory advisors with the responsibility for advising on development impacts have a maximum threshold against which they consider cumulative impacts, no such limits have been stated (and indeed there may be no firm limits, see e.g. Natural England 2015). In the absence of guidance on acceptable or tolerable thresholds the only reliable guide appears to be the most recent cumulative total for a consented development. Thus, a conservative headroom estimate can be derived in relation to this cumulative consented total, although subsequent changes in the knowledge of the status of the seabird populations in question may also affect this.

In the UK, Collision Risk Modelling (CRM) methods have become largely standardised, simplifying interpretation of the results obtained. In order to ensure that potential impacts are not underestimated wind farms are assessed on the basis of the worst case scenario (WCS) in terms of predicted numbers of collisions. This is usually represented by the largest number of small dimension turbines which could be installed (i.e. within the range of options under consideration by the developer at the time of the assessment). Wind farm assessments are also required to take into account the potential cumulative mortality across all wind farms which may affect the same seabird populations. The cumulative totals for each species are made up of the WCS mortality for each contributory wind farm, taken either from the wind farm Environmental Statement (ES) or the Development Consent Order (DCO). Wind farm alone mortality is rarely considered to be of concern for a wind farm in isolation. However, cumulative and in-combination totals (for Habitats Regulations Assessment, HRA, in relation to SPAs) are often subject to considerable scrutiny during the assessment and consenting process.

Constructed offshore wind farms, particularly more recent ones, rarely use the number or type of turbines detailed in the application. Technological developments mean that generating capacities can be attained with fewer, larger dimension turbines. Collision mortality is almost always lower for these 'as-built' developments when compared with consented designs. Re-calculating collision mortality for built wind farms with updated parameters has the potential to reduce the predicted mortality, thereby increasing collision headroom. This is straightforward for constructed wind farms, however consented but as yet unbuilt or partially built wind farms may not have reached a final determination on turbine model (and number) making updates potentially less reliable. However, at such sites it is likely that the smaller consented turbines will have subsequently been ruled out in favour of a smaller number of larger turbines. Thus, updates can also be applied to these developments, albeit with the proviso that this remains indicative and further calculation may be required in future once the final design is known and/or fixed.

The Crown Estate (TCE) is interested in identifying the magnitude of headroom available within the consented offshore wind portfolio for key seabird species, which could be translated into future wind farm developments. The methods for undertaking this calculation were developed by MacArthur Green in a previous piece of work conducted for TCE. This work found that, owing to the structure of the Band collision model, the original mortality could be updated using a simple equation relating old to new turbine parameters (MacArthur Green 2016). This report presents the results of the application of this approach to UK offshore wind farms and accompanies a spreadsheet (*Ornithology CRM Headroom TCE 13_01_2017.xlsx*) which contains the data used, the calculations and results obtained for five seabird species: gannet, kittiwake, lesser black-backed gull, great black-backed gull and herring gull. These species were selected on the basis that they are found widely around UK coasts in most months and spend a significant proportion of their time at potential collision height (i.e. >22m), with the consequence that they usually have the highest collision risk estimates in offshore wind farm assessments.

This work builds on a project which MacArthur Green undertook for the Marine Management Organisation (MMO) and TCE which was developed as part of the offshore wind industry's Coping Strategy. That project (title: '*Ornithological data inventory for offshore wind farm consenting*') involved collating publicly available data relating to offshore wind farm collision estimates.

3. Methods

The method for recalculating collision mortality uses ratios of consented and built turbine parameters to adjust the consented mortality estimates and does not require re-running of the collision model. The wind farm parameters required (both those on which the original assessment was based and updated ones for the built wind farm) are:

- Number of turbines;
- Rotor radius;
- Blade pitch;
- Max blade width (chord); and,
- Average RPM.

These data were sought for all UK offshore wind farms and were entered into a table in Excel ('Wind Farm Specifications'). Having collated these data there are two steps required for re-calculating collision mortality.

1. Calculate the species-specific probability of collision for a single transit for old and new turbine specifications; and,
2. Calculate the adjusted mortality using Equation (1).

$$\boxed{\textit{Updated mortality} = \textit{Original mortality} \times (r_0/r_1) \times (trf_1/trf_0) \times (p.\ collision_1/p.collision_0)} \qquad [1]$$

Where:

r = rotor radius

trf = total rotor frontal area (rotor area x no. of turbines)

p.collision = probability of collision on single transit (derived from the Band model)

and,

subscript *0* = original value,

subscript *1* = updated value.

[Note that the radius ratio is original/updated, while the other two terms are updated/original].

To undertake step 1, calculation of the probability of collision for the two turbine specs at each wind farm, the '*Single transit collision risk*' tab as presented in the Band CRM Excel spreadsheet was used. To facilitate later calculations the original layout of this sheet in the Band model was rearranged so that all the data and calculations for any given wind farm were contained in a single column. This made it possible to incorporate the calculations for all wind farms in a single sheet, greatly simplifying the presentation of the calculations.

As the probability of collision is also species specific, a separate sheet was created for the five species (e.g. Gannet p.collision ratio, etc.). The turbines parameters used in the calculation are the rotor radius, blade pitch, blade width and rotor rpm, while the bird parameters are bird length, wing span, flight speed and flight type (gliding or flapping). The outputs for each species were the probability of collision based on the two turbine specifications and the ratio of the two.

Step 2, using Equation (1), was then undertaken in a further sheet (CRM recalculation) which presents the original and updated mortality estimates and the difference between the two for all wind farms. It should be noted that the original mortality has been presented with recent revisions to the avoidance rate already applied. The avoidance rate revisions followed a review of seabird collision monitoring studies conducted by the BTO on behalf of Marine Scotland (Cook et al. 2014), which recommended increased collision avoidance rates for gannet, kittiwake, lesser black-backed gull, great black-backed gull and herring gull. The statutory nature conservation bodies subsequently issued a joint guidance note (JNCC 2014) which accepted (with minor modifications) the recommendations in Cook et al. (2014). Subsequent wind farm cumulative assessments have applied these avoidance rate adjustments retrospectively to all the wind farms included in their assessments. Thus, for large gulls the mortality presented in older wind farm assessments was typically calculated with a 98% avoidance rate (in conjunction with the model Option 1), which has been updated using the current accepted rate of 99.5% for these species and this model. As this four-fold reduction has already been applied in recent wind farm assessments (e.g. for East Anglia THREE) it is therefore appropriate to apply the same adjustment prior to applying wind farm based recalculation to estimate the headroom available.

3.1 Bird parameters

Although the recalculation is based primarily on changes to the turbine parameters it is necessary to include bird biometric estimates (e.g. body length) to calculate the change in the probability of collision for a single rotor transit (‘p.collision’). It is current best practice is to present these as part of the collision risk assessment, and this has helped ensure that a standard set of values are used. These values were used for the recalculation and are presented in Table 1.

Table 1. Species biometrics used for the recalculated probability of collision (p.collision).

Species	Body length (m)	Wing span (m)	Flight speed (ms^{-1})
Gannet	0.94	1.73	14.9
Kittiwake	0.39	1.08	13.1
Lesser black-backed gull	0.58	1.42	13.1
Great black-backed gull	0.71	1.58	13.7
Herring gull	0.60	1.44	12.8
Sandwich tern	0.39	1.00	10.5

If original collision assessments reported the parameter values used and these were different from those in Table 1 then these have been used to calculate the original p.collision value. If no values could be found in the documentation then the current values (Table 1) have been assumed. Any future change to these values (if advocated by the SNCBs) , if applied retrospectively, would lead to further collision estimate revision.

4. Caveats and Assumptions

The headroom calculations are based on Band model Option 1 or 2 (Band 2012). These are the basic versions of the collision model, which make no allowances for the relationship between flight height and collision risk but rather assumes a fixed proportion of birds fly at rotor height (PCH – potential collision height). The proportion will typically have been either estimated from the site-specific surveys conducted for the impact assessment (Option 1), or from a dataset pooled across many sites (Option 2; using data from e.g. Johnston et al. 2014). The latter approach is typically used when site specific surveys recorded flight height estimates for relatively few individuals of a given species. It is straightforward to apply a revision for changes in the PCH (multiply by the ratio of old to new PCH). However, estimates of PCH (and also what height bands were used to define rotor heights during surveys) and the sample size for the estimate are not consistently provided in assessments. Consequently, the current headroom calculations do not apply this adjustment. While there is scope to add this at a later stage, since the lower rotor tip height is restricted by the needs of shipping (e.g. 22m above HAT) and raising turbines adds considerable expense, it seems likely that most wind farms will have been constructed with the minimum permissible clearance and this will also have been the basis for the assessment. It is therefore unlikely that there will be much gain in headroom from this adjustment (certainly when compared with the gains from reduced turbine numbers).

For a small number of consented (but not yet constructed) wind farms the results from Band model Option 3 were accepted for the impact assessment and these value have not subsequently been revised for the basic model (this applies to the Dogger Bank projects). This model estimates collisions using seabird flight height curves which account for the fact that the density of seabirds in flight (generally) decreases with height. As the risk of collision also varies between the rotor tip and hub, these two relationships are combined to obtain collision estimates which reflect the fact that most seabird flight activity is at zero risk of collision (i.e. below rotor height) and those flights which do overlap with rotor heights are at low risk of collision as they occur within the outer section of the rotors. Option 3 collisions cannot be updated using the method applied here as part of the calculation includes estimation of the overlap between the bird height distribution and turbines. Furthermore, questions remain about the reliability of some of the data used to derive the seabird flight height curves and thus this approach has not been universally adopted by statutory advisors. As a result, the current preference of the statutory agencies is for collision assessments to be based on Options 1 or 2. Therefore, for those wind farms which were consented on the basis of Option 3, it has been necessary to review the technical reports in order to obtain collision mortality estimates derived using Options 1 or 2. Thus, for these sites the consented collisions may differ from the values used here, and consequently there are differences between the cumulative total estimated for this project and recent cumulative impact assessments. For example, the final East Anglia THREE cumulative total for gannet mortality at UK wind farms in the North Sea and English Channel was 2,874 (EATL 2016), whereas the equivalent estimate using Options 1 and 2 as estimated here is 2,999.

Not all the turbine parameters used in the recalculation are straightforward to obtain. For most wind farms the number of turbines and the rotor dimensions can be obtained. If the turbine model can be identified then the maximum blade width can be obtained from the manufacturer's specifications. The remaining parameters (average blade pitch and average RPM) proved generally harder to obtain. To fill in these gaps, a range of options was adopted. If values for the missing parameter were available for the same turbine model used elsewhere these were used. The same approach was applied if the turbine model was considered likely to share characteristics (e.g. listed as the same dimensions for rotor diameter, but a different rated MW output). A similar approach was adopted if the turbine model was not known but the dimensions matched those for another site or turbine model. RPM was often given as a maximum and minimum with no average. If this was the case the maximum was used as this is the more precautionary. In one or two cases it was possible to back-calculate missing turbine parameters using known ones and the probability of collision. The least often available parameter was blade pitch. This value is often also difficult to obtain for wind farm assessments. In the latter cases a default of 15° is used and this default value was also used here.

The proportion of flights recorded at Potential Collision Height (PCH) is another parameter used in the mortality calculation. For older wind farms this was typically obtained through the assignment of birds in flight to broad height bands (e.g. 0-20m, 20-150m, 150m+), with the number of flights in the middle band used to estimate the proportion at collision risk height. The use of these bands was necessary since surveys are commissioned prior to final decisions on turbine model and hub height have been made. In some cases the PCH value was then adjusted for assessed rotor heights (e.g. if the proposed rotor spanned 25 to 150m the survey estimate would be recalculated using 125/130 = 96%). More recent applications, using digital aerial surveys, have calculated PCH on the basis of individual flight height estimates, however the method is essentially the same.

While updating the original PCH values for ones more closely matching the dimensions of installed turbines, the required data were rarely provided to permit this. It may be possible to investigate this aspect further, however following a preliminary review this was considered to be a low priority for three reasons:

1. As noted above, PCH has been calculated in a variety of methods which are often not explained in sufficient detail to permit recalculation. It was estimated that fewer than 10 wind farms had sufficient data;
2. When adjustment did appear to be possible, the change in lower tip height was small, generating mortality changes in the order of <5%; and,
3. Because the original methods for calculating PCH have varied and are often not explained clearly, this aspect would require a considerable amount of additional documentation in order to provide the necessary evidence base expected to satisfy SNCBs (in contrast with the

turbine information used for the main CRM adjustment which is much more transparent and less open to debate and confusion).

Consequently, no adjustment for PCH has been included here. If this is undertaken in the future it is recommended that a detailed summary of the data and methods used for each wind is provided in order to maximise SNCB acceptance.

5. Results

The Excel file contains several sheets, named as follows (note some species names have been abbreviated due to the character limit on tab names):

- Guide
- Wind Farm Specifications
- CRM recalculations
- CRM per MW
- NNF SPA CRM recalculation
- R&A SPA CRM recalculation
- MB SPA CRM recalculation
- BF SPA CRM recalculation
- AOE SPA CRM recalculation
- FFC pSPA CRM recalculation
- Gannet p.collision
- Kittiwake p.collision
- Lssr. BB gull p.collision
- Gt. BB gull p.collision
- Herring gull p.collision
- S. tern p.collision

The following sections provide a guide to the contents of each sheet. The Excel file has been constructed with links between sheets so that if new information becomes available this can be entered in to the appropriate cells (primarily Wind Farm Specifications) which will automatically update the CRM adjustment and update mortality estimates.

5.1 Wind Farm Specifications

This sheet contains a summary of the collision parameters for each UK offshore wind farm. The key parameters are those relating to the turbine specifications. As noted above, these were not all available for all sites, however in most cases values could be estimated from other locations. This table also includes turbine hub height, however this value is not currently used in the calculations (see above re the use of Band model Option 3). Individual cells have additional notes to provide supporting information (e.g. on turbine model, etc.).

5.2 CRM recalculations

This sheet lists all the UK offshore wind farms included in the assessment for each species. Wind farms are assigned to three regions (E, W, S), followed by the original and updated values for the rotor radius, total rotor frontal area (TRF) and probability of collision (P.collision), extracted from the preceding sheets. These values are then used to obtain an overall CRM adjustment value (a value by which the original mortality can be multiplied to obtain the updated mortality) using equation 1.

The most recent estimate of the consented mortality for each species is listed in the next column. These were extracted from the East Anglia THREE wind farm cumulative assessment for the North Sea and English Channel sites, and from the individual wind farm assessments for Irish Sea sites (East Anglia THREE is the most recent development to be examined by the Planning Inspectorate and Natural England accepted the cumulative collision totals presented in the assessment). The Band model used for the reported mortality is given (either 1 or 2) and confirmation of the avoidance rate applied (in all cases 98.9% for gannet and kittiwake and 99.5% for the large gulls). The next two columns provide the recalculated mortality, following application of the CRM adjustment, and the headroom (the difference between the original and updated mortalities). The final columns provide the total and regional headroom summaries. These are also presented in Table 2. For all species, the larger number of wind farms in the North Sea means that the majority of the collisions (and hence headroom) are accounted for here.

A column of the current mortality (revised figure where data are available or original if no update is possible) divided by the wind farm capacity (in MW, the most recent estimate available) is provided at the right-hand end of the table. These data are analysed further in the 'CRM per MW' sheet.

5.3 CRM per MW

Summary statistics for the collisions per MW estimates have been provided by region (East, West and All). Further discussion on the results is provided later.

5.4 Species p.collision

These sheets have been adapted from the 'single rotor transit collision' sheets in the Band model Excel file. While the latter presents the various calculations in tabular form, here the tables have been rearranged into columns to permit calculation for each wind farm in a single sheet, with the calculations repeated for the two sets of turbine parameters. The results of the calculations are provided in the first 12 rows, below which are the turbine parameters (linked to the Wind Farm Specification sheet), the species biometrics used (body length, wing span, flight speed, etc.) and then the cells which perform the calculations.

Table 2. Summary cumulative collision mortality for UK offshore wind farms, and split between those in the North Sea (including English Channel) and those in the Irish Sea. Original collision estimates are those for consented projects, updated estimates have been recalculated using as built (or planned) wind farm specifications and headroom is the difference between the two.

Species	UK Collision Mortality Total			UK North Sea Collision Mortality Total (inc. English Channel)			UK Irish Sea Collision Mortality Total		
	Original	Updated	Headroom	Original	Updated	Headroom	Original	Updated	Headroom
Gannet	3055	2618	**437**	2999	2589	**409**	56	28	**28**
Kittiwake	3949	3288	**661**	3726	3172	**554**	223	116	**107**
Lesser black-backed gull	682	412	**270**	503	302	**201**	179	110	**69**
Great black-backed gull	913	637	**276**	885	623	**262**	28	14	**14**
Herring gull	803	521	**282**	724	479	**245**	78	41	**37**

5.5 Assignment of mortality to SPAs

In addition to collision risk modelling for their impact assessments, wind farms within foraging range of SPA breeding colonies have also typically undertaken Habitats Regulations Assessments (HRA) for those species considered at risk of likely significant effects. In most cases these assessments cover relatively small areas (reflecting individual species' foraging ranges) and are focussed on the breeding period as this is when impacts are considered most likely.

The key species at risk of collisions in English waters for which offshore wind farm HRA has been undertaken are gannet, kittiwake, lesser black-backed gull, herring gull and Sandwich tern. The estimated number of individuals of these species predicted to be at risk of collisions from SPAs at which they are designated features have been provided in named sheets in the Excel spreadsheet, with the updated mortality and headroom calculated. The abbreviated names for each SPA and the species included are:

- NNC - North Norfolk Coast SPA (Sandwich tern)
- R&A – Ribble and Alt Estuary SPA (lesser black-backed gull)
- MB – Morecambe Bay SPA (lesser black-backed gull, herring gull)
- BF – Bowland Forest SPA (lesser black-backed gull)
- AOE – Alde Ore Estuary SPA (lesser black-backed gull, herring gull)
- FFC – Flamborough and Filey Coast pSPA (gannet and kittiwake)

Sandwich tern has not been assessed more widely for this project as their distribution and foraging habits are generally quite localised with the consequence that (with notable exceptions) they have rarely been considered at significant risk of collision impacts. However, there are particular regions where this species may be identified as a key risk, hence its inclusion in the SPA section.

Furthermore, this list omits great black-backed gull because the only SPA for this species with breeding season connectivity to English waters is the Isles of Scilly. Therefore, HRA will only be required in relation to this species for wind farm proposals within 60k of these islands (note also that this is likely to be an over-estimate: 60km has been derived from the herring gull estimate which has been used due to the very limited data for great black-backed gull).

For most of the HRA conducted for these SPAs, the focus of interest has been the breeding season, since this is the period for which the species are designated, with assessment of collisions during the breeding months at wind farms within foraging range. For these reasons the excel sheets only include a subset of wind farms. Appendix 1 provides maps for lesser black-backed gull, herring gull and Sandwich tern which show the foraging ranges from their SPAs which overlap (or have the potential to overlap) with offshore wind farms. For the SPAs included in the Appendix 1 figures (A1.1 to A1.3) there has been no systematic attempt to attribute nonbreeding mortalities to the populations. However, gannet and kittiwake from the FFC pSPA have received more attention with regards to potential impacts, including estimation of the proportion of nonbreeding season collisions which can be attributed to these populations, and thus they are discussed in more detail below and maps are included in the text.

The gannet and kittiwake mortality estimates attributable to the Flamborough and Filey Coast pSPA have been taken from the East Anglia THREE in-combination assessment (EATL 2016) as this is the most recent HRA available. For these SPA features original and updated mortality has been split into three seasons: breeding, post-breeding and pre-breeding (Furness 2015). The proportion of mortality attributed to the FFC pSPA populations in each season is based on estimates of the foraging ranges in the breeding season and studies of migration for the nonbreeding seasons (see MacArthur Green 2015 for details – these reports are included in Appendices 2 and 3). The nonbreeding apportioning method is based on studies of migration in these species, which provides a guide to movements post- and pre- breeding and hence the wind farms which may be encountered. Table 3 summarises the original mortalities, the updated values and the estimated headroom. Figures 1 and 2 show the breeding season foraging ranges from FFC pSPA for these species (plus other British SPAs which could be connected to offshore wind farms in English waters) and the estimated percentage of total mortality in the nonbreeding seasons at current wind farms which can be attributed to the FFC pSPA populations.

Table 3. Seasonal and annual cumulative collision mortality for Flamborough and Filey Coast pSPA populations of gannet and kittiwake at UK offshore wind farms. Headroom is the difference between the original and updated collision estimates.

Species	Breeding			Post-breeding			Pre-breeding			Annual		
	Original	Updated	**Headroom**	Original	Updated	**Headroom**	Original	Updated	**Headroom**	Original	Updated	**Headroom**
Gannet	162	110	**52**	21	14	**7**	15	11	**4**	198	135	**63**
Kittiwake	165	145	**20**	73	61	**12**	81	73	**8**	319	279	**40**

For gannet over 80% of the mortality for the FFC pSPA population is predicted to occur during the breeding season. This reflects two aspects of this species biology: very few individuals from UK colonies remain in the North Sea during winter and large numbers of individuals from other colonies pass through the North Sea on migration. This greatly reduces the proportion of gannet collisions (during the non-breeding season) which are expected to be breeding birds from the FFC pSPA. The implications of this for future wind farm development is that new projects within gannet foraging range of the FFC pSPA would be expected to account for most of the headroom (assuming the assessment method remains as used for East Anglia THREE). However, as can be seen on Figure 1, gannets are not uniformly distributed within the foraging radius of the colony. Thus, fewer collisions would be predicted at locations in lower density areas (e.g. to the south of the Hornsea zone) than at locations closer to the colony (e.g. immediately to the east of the colony). It should also be noted that presenting mortalities for individual colonies in this manner (as required for HRA) clearly omits important aspects of the species' biology and distribution. In this case, the gannet densities increase to the north of FFC pSPA, due to the increasing proximity to the largest breeding population of gannets in the world, at the Bass Rock. Furthermore, the broad divisions of breeding season mortality at wind farms in the North Sea indicated on Figure 1 reflect the contribution of birds from the Bass Rock to the total collision estimates, rather than the fact that FFC pSPA birds don't forage in this area (e.g. gannet breeding season collision mortality is attributed equally between Bass Rock and FFC pSPA birds at the Dogger Bank wind farms, even though these wind farms are similar distances from FFC pSPA as wind farms to the south). On this basis, while a wind farm off the coast of Northumberland would be expected to contribute fewer FFC pSPA collisions than one off Humberside, the former would also need to be assessed against the Bass Rock colony, while the latter would only be assessed against FFC pSPA. And overall, the gannet densities suggest that the wind farm off Northumberland would have higher gannet mortality. It is also important to remember that although 100% of breeding season collisions at wind farms in the outer Thames would be assigned to FFC pSPA, in reality there are very few predicted gannet collisions at these locations farms at this time of year and this assignment is precautionary.

The results for kittiwake are somewhat different, with around 50% of annual mortality predicted to occur in the breeding season and 25% in each of the two nonbreeding seasons. This reflects the smaller contrast in predicted FFC pSPA collision mortality across seasons for most North Sea wind farms (i.e. 17% in the breeding season and 5% to 7% in the nonbreeding seasons) which in turn is a reflection of the smaller foraging range for this species (60km compared with 229km for gannet).

Only two wind farms are located within foraging range from the FFC pSPA: Humber Gateway and Westermost Rough, neither of which had high kittiwake mortality predictions. While this could appear to imply that wind farm location is less important for this species, it is apparent from the summer densities (Figure 2) that the region to the east and north-east of FFC pSPA has the highest densities and thus wind farm development within this area will account for more of the available headroom than equivalent locations to the south-west of the pSPA. It should also be noted that, as for gannet, there are more breeding colonies to the north of FFC pSPA, which also account for the higher densities off the Northumberland coast. In contrast, there are very few kittiwake colonies, and none with more than a few tens of pairs, to the south of FFC pSPA.

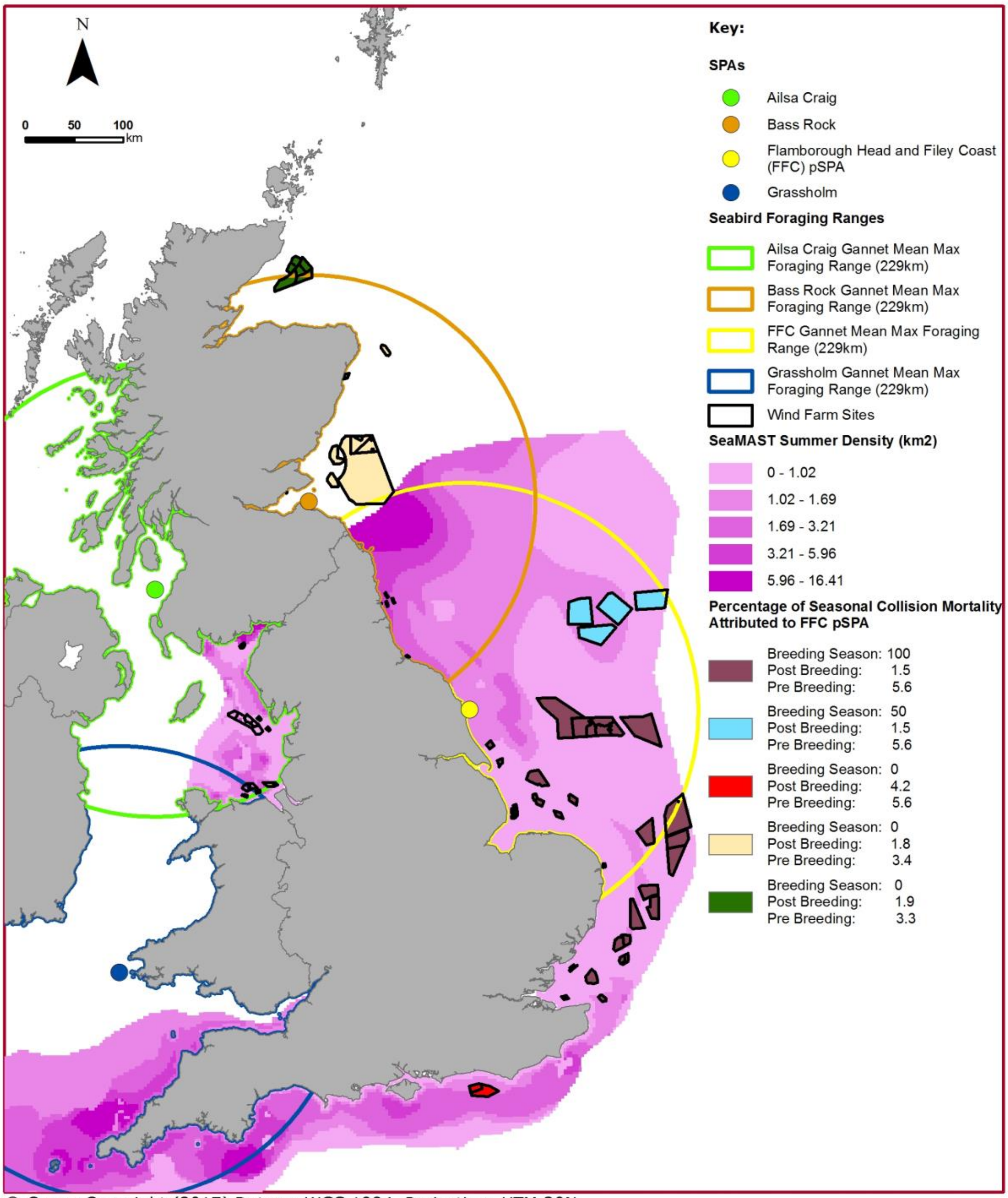

Figure 1. Density of gannets in summer (SeaMAST output) overlaid with mean maximum foraging range from key SPAs with potential for connectivity to wind farms in English waters. Wind farms are colour-coded to identify different contributions to seasonal collision mortality of the FFC pSPA population.

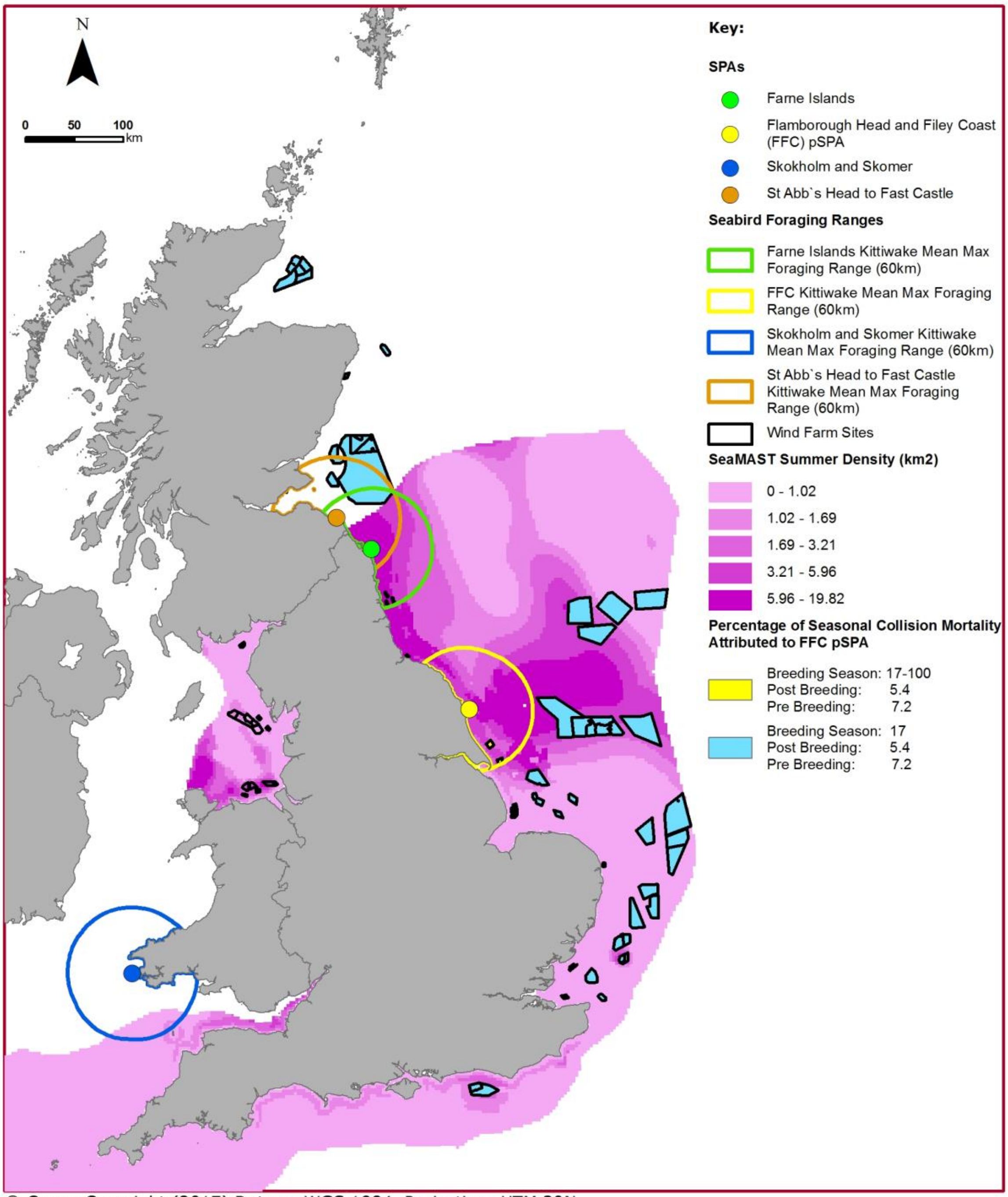

Figure 2. Density of kittiwakes in summer (SeaMAST output) overlaid with mean maximum foraging range from key SPAs with potential for connectivity to wind farms in English waters. Wind farms are colour-coded to identify different contributions to seasonal collision mortality of the FFC pSPA population.

6. Collision mortality per MW and implications for future offshore wind development

In order to further understand how estimated headroom translates into potential wind farm capacity, the estimated mortality per megawatt has been calculated at each wind farm (see 'CRM per MW' sheet in the excel file). The intention is to provide a guide to how much additional wind power capacity could be installed before the current mortality threshold is reached (i.e. the level of the most recent cumulative assessment). These estimates have been summarised for east (North Sea) west (Irish Sea) and combined.

There is a considerable variation in collision mortality per megawatt capacity among wind farms (Table 4, Figure 3). At most wind farms, collision mortality per megawatt is very low (using the updated mortality estimates), with a median estimate below 0.04 collisions/MW for all species. There are only 10 species & wind farm combinations where the collisions per megawatt estimates are greater than 0.5:

- Blyth (for all species except lesser black-backed gull),
- Firth of Forth Alpha and Bravo (gannet and kittiwake),
- Neart na Gaoithe (gannet), and
- Teesside (kittiwake).

The median and mean collision mortalities per megawatt for gannet and kittiwake are an order of magnitude higher than for the large gulls. Therefore, in terms of providing guidance for potential future wind farm capacity, gannet and kittiwake would be expected to be the species most likely to set future limits. Considered across English waters as a whole, the mean collision estimates per megawatt (which are higher than the median values and therefore more precautionary) for gannet and kittiwake are 0.15 and 0.14 respectively. On the basis of the total headroom calculated above (Table 2) these generate estimates of additional wind farm capacity of approximately 2,880 MW and 4,714 MW respectively. However, these estimates are not evenly split across regions: the North Sea mortalities per MW are much higher, at 0.2 and 0.18 for gannet and kittiwake respectively. This indicates that the North Sea additional wind farm capacity would be around 2,000 MW (gannet) or 3,070 MW (kittiwake).

Since limits on development would be defined by the lower value, this indicates that on the basis of the current increase in headroom, and assuming new development is located in 'average' positions with respect to collision risk, the current wind farm portfolio could be expanded by around 10% (for the wind farms used in this project the total generating capacity is estimated to be approximately 23,000 MW). Clearly, if new wind farms are located such that gannet mortality is predicted to be above average (e.g. in the higher density areas indicated on Figure 1) then this would be expected to reduce the potential for expansion, and vice versa.

There is a greater range in the per MW estimates for kittiwake than gannet (Figure 3). This is likely to be a reflection of the differences in foraging ranges for these two species. Gannet forage over much wider areas with the consequence they are recorded widely at relatively consistent (and potentially lower) densities. In contrast, kittiwakes forage over much shorter distances which results in relatively high densities near colonies, and lower densities between colonies. Furthermore, kittiwake are likely to be more coastal than gannet, and there is therefore a greater likelihood of overlap with earlier rounds of wind farm development which are found in shallower near coast waters. Thus, while gannet mortality per MW is comparatively low and even across most wind farms, kittiwake mortality per MW is slightly higher at nearshore, close to colony wind farms with the result that the overall per MW mortality is higher for this species.

For gannet and kittiwake which are found widely, larger scales are appropriate as the wide variability in estimates across wind farms means that it is very difficult to reliably refine predictions to finer spatial scales. However, the SeaMast density outputs (e.g. Figures 1 and 2) provide a guide to areas which are likely to have higher or lower collision risks. However, for other species finer scale assessment is more straightforward and is likely to be of greater interest. For example, within the foraging range of Sandwich terns from the North Norfolk Coast SPA there are four wind farms for which mortality has been estimated (Dudgeon, Race Bank, Sheringham Shoal and Triton Knoll). The per MW mortality at the three nearest wind farms is very similar (c. 0.04), while the estimated headroom is 35. This indicates that, on the basis of this species alone, within this area there may be potential for up to 875MW of additional wind capacity. As for gannet and kittiwake, the SeaMast outputs provide a useful additional guide on areas likely to be of higher or lower risk (Appendix 1).

Table 4. Summary annual collision mortality (using updated estimates) per MW of wind farm generating capacity, split into E (North Sea) W (Irish Sea) and All regions (for wind farm list see accompanying spreadsheet).

Summary statistic	Region	Gannet	Kittiwake	Lesser black-backed gull	Great black-backed gull	Herring gull
5% percentile	E	0.000	0.000	0.000	0.000	0.000
Median		0.041	0.058	0.003	0.013	0.000
Average		**0.197**	**0.182**	**0.016**	**0.083**	**0.052**
95% percentile		1.116	0.760	0.073	0.247	0.306
5% percentile	W	0.000	0.000	0.000	0.000	0.000
Median		0.000	0.000	0.000	0.000	0.000
Average		**0.005**	**0.017**	**0.035**	**0.002**	**0.008**
95% percentile		0.031	0.101	0.114	0.008	0.044
5% percentile	All regions	0.000	0.000	0.000	0.000	0.000
Median		0.017	0.034	0.003	0.003	0.000
Average		**0.147**	**0.139**	**0.021**	**0.062**	**0.043**
95% percentile		0.926	0.688	0.114	0.148	0.208

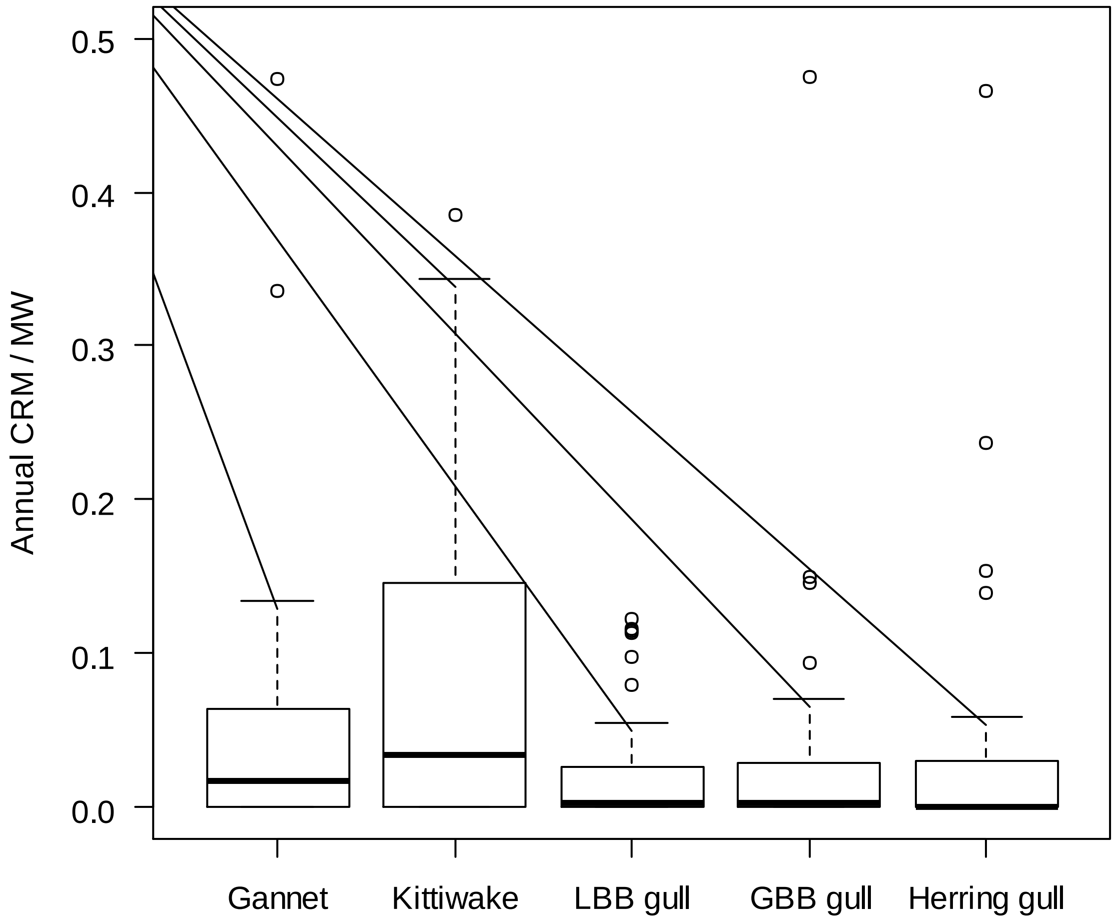

Figure 3. Box plot of annual collision mortality (updated values) per megawatt. Thick lines are the median value, boxes indicate the 25% and 75% range and dashed lines the upper 95% range. The y-axis has been cropped at 0.5 to improve clarity. This omits four outliers for gannet (0.7, 1.0, 1.3, 2.1) and kittiwake (0.6, 0.7, 0.9, 1.3), one for great black-backed gull (1.6) and one for herring gull (0.7).

7. Population status and implications for headroom acceptance

As discussed above, the concept of collision headroom is dependent on two factors; an estimate of the cumulative mortality and a tolerable mortality threshold. While calculation of the first of these is relatively straightforward and can also be updated as new wind farm information becomes available, the latter values have not been stated by regulators or SNCBs for any species. In the absence of guidance from statutory agencies on how they determine tolerable limits it is necessary to infer what these may be from comments provided on wind farm cumulative assessments. Hence, for the purposes of this work we have assumed the most recently submitted cumulative mortality represents a precautionary estimate of the threshold (at the time of writing this is for the East Anglia THREE wind farm which has not yet been consented, but for which NE accepted the mortality estimates for the key collision risk species: gannet, kittiwake and the three large gulls).

However, even this cannot be considered to represent a fixed threshold, since the status of the seabird populations and our understanding of their ecology and factors affecting their demography will affect the magnitude of mortality which will be considered acceptable. With this in mind, the following sections review the population data and status for the five key species focussed on this report.

7.1. Gannet

About three-quarters of the world's gannet population breed in Britain and Ireland, so we have a particularly strong responsibility to monitor and protect this species. Following censuses of the world's gannet colonies in 1900, 1939, 1949, 1969-70, decadal counts at colonies in Britain and Ireland have been made since the 1980s (Wanless et al. 2005). Because gannets are relatively easy to census (now by aerial photography), there is high confidence in the count data. From 50,000 pairs (or Apparently Occupied Nests AONs) in 1900, numbers increased slowly until 1939, then more rapidly at about 2% per annum up to 1994-95. From then to 2003-04 the rate of increase fell to 1% p.a., with 261,000 AONs in Britain and Ireland in 2003-04 (Wanless et al. 2005), the lower growth rate possibly indicating density-dependent constraints starting to act. However, the Scottish population increased by a further 33% from 2003-04 to 2013-14 (Murray et al. 2015). Small colonies have tended to grow faster than large colonies, and several new colonies have formed as the population has grown. The Bass Rock colony has become the largest gannet colony in the world with 75,259 AONs in 2014 (Murray et al. 2014).

Continued growth of gannet colonies may have been favoured by the recent increases in abundance of herring and mackerel stocks, on which gannets feed extensively. Climate warming and the northward spread of mackerel has been suggested as the cause of recent establishment of a successful gannet colony as far north as Bear Island (74°27' N) in the Svalbard Archipelago (Anker-Nilssen et al. 2016). Reductions in amounts of fish discarded by fisheries may reduce food supply for gannets, especially in winter, but it is uncertain whether gannets depend on discards or simply feed on them when available.

Breeding success of gannets is consistently very high, suggesting that they have more than adequate food supplies during the breeding season, although birds from larger colonies tend to travel further for food while breeding which suggests some competition.

Most gannet colonies in Great Britain are included in the Natura2000 network as SPAs for breeding gannets so that 96% of the GB population is represented within SPAs (Stroud et al. 2016). The only gannet colony in England, at Flamborough & Filey Coast pSPA, which is the closest gannet colony to many UK offshore wind farms, has increased from 720 pairs in 1986 to 3,940 in 2004, 7,859 in 2009, 11,061 in 2012 and 12,494 in 2015 (JNCC SMP online database).

In summary, the good conservation status of gannets would suggest that the population will be relatively resilient to impacts from offshore wind collision mortality, but due to the concentration of wind farms in the North Sea, the colonies at greatest risk of impact will be at Flamborough & Filey Coast pSPA and Bass Rock (Forth Islands SPA). The main concern relates to in-combination and cumulative impacts of collision mortality, with particular focus on HRA concerns about in-combination impacts on the population at Flamborough & Filey Coast pSPA and Bass Rock (Forth Islands SPA).

General options for further increasing gannet collision mortality headroom (i.e. measures which would be effective across most wind farms) include refining flight height estimates using altimeter data, deriving evidence based estimates of nocturnal flight activity and conduct further work on avoidance rates. Altimeter deployments on adult gannets foraging from the Bass Rock (Cleasby et al. 2015) have revealed behaviour based differences in flight height, with commuting flight generally lower (below PCH) than active foraging flight. It would be useful to extend this work to nonbreeding periods, although longer term deployments would probably introduce complications due to the need to regularly calibrate pressure operated altimeters. Data to refine nocturnal flight activity have already been collected by other research projects so this would be fairly straightforward to conduct once the data were obtained. Derivation of an evidence-based avoidance rate for gannet should be possible from analysis of the data collected for the ORJIP project. This is likely, but not certain, to give an avoidance rate that is higher than the value recommended by NE for existing CRM calculation.

Adults foraging from the colony at FFC pSPA have been tracked by the RSPB (e.g. Langston et al. 2013). While these data have been reported on by the RSPB, they have not been used to estimate the magnitude of breeding season connectivity of this colony with existing and planned wind farms. Such analysis may be planned or underway by the RSPB, or could be conducted by other researchers if these data could be obtained. The habitat utilisation maps which could be developed would allow refinement of the extent of connectivity with existing wind farms (hence could reduce the number of collisions attributed to FFC pSPA) and would also permit identification of areas of higher and lower risk for future wind farm development. Given that commuting flight appears to be of lower collision risk than active foraging flight, a better understanding of the areas used for foraging would clearly be of great value. Furthermore, as tracking has been conducted over several years it should be possible to determine the extent of consistency in site selection between years.

7.2. Kittiwake

Kittiwake breeding numbers in the British Isles increased considerably from 1900 to the 1980s (Coulson 2011). However, breeding numbers have since decreased, with strongest decreases in Shetland (for example on Fair Isle from 20,000 pairs in 1987 to 2,000 pairs in 2014 (Fair Isle Bird Observatory Report for 2014), while breeding numbers declined by 93% at Noss and by 86% at Foula between 2000 and 2015 according to the JNCC SMP database). Numbers decreased by 66% in Scotland as a whole from 1986 to 2011 (Foster and Marrs 2012) but have remained approximately stable in recent years at the largest colony in England (Flamborough & Filey Coast pSPA; 42,692 AONs in 2000, 45,278 AONs in 2016 JNCC SMP database). It should be noted that although there has been some disagreement between wind farm developers and the SNCBs about the reliability of some of the historical counts at FFC pSPA, the broad trends in kittiwake colony counts over the last two decades are not in dispute.

Kittiwakes at North Sea colonies feed mainly on sandeels, and show reduced breeding success when sandeel stocks decline (Oro and Furness 2002, Frederiksen et al. 2004). Kittiwakes are considered to be particularly sensitive to impacts of increased sea temperatures through the effect of those on sandeel abundance (Frederiksen et al. 2004), but in addition are affected by increased sea temperatures outside the breeding season in wintering areas (Laffoley and Baxter 2016). Kittiwakes are also subject to increased predation impacts when food availability for large predatory seabirds is reduced (Oro and Furness 2002).

In line with observed trends and ecological relationships in Scotland, Sandvik et al. (2014) predicted that kittiwakes would be extirpated from Norwegian breeding colonies within 10 to 100 years as a consequence of increasing sea temperatures and altered marine ecosystems, and the current trend in breeding numbers in Norway is consistent with that prediction (Anker-Nilssen et al. 2016).

Breeding numbers at the Isle of May (Forth Islands SPA) have been approximately stable in recent years. That colony is subject to detailed monitoring of kittiwake ecology by CEH scientists, and provides a sentinel of human impacts on the marine environment in relation to climate change, fisheries and offshore renewables. In the UK, the relative importance of the Flamborough & Filey Coast pSPA kittiwake population has increased as it has maintained moderately good breeding success and breeding numbers in recent years while numbers in much of northern Britain have declined towards extinction. As a result, there is likely to be especially strong emphasis on ensuring that the Flamborough & Filey Coast pSPA kittiwake colony remains at good conservation status. However, the trend in kittiwake numbers predates wind farm development and there is a strong case to be made that, while wind farms may cause additional mortality, the primary driver of current declines lies elsewhere.

The main option for further increasing kittiwake collision mortality headroom (i.e. measures which would be effective across most wind farms) would be to derive evidence based estimates of nocturnal flight activity and conduct further work on avoidance rates. Data to refine nocturnal flight activity have already been collected by other research projects so this would be fairly straightforward to conduct once the data were obtained. Derivation of an evidence-based avoidance rate for kittiwake should be possible from analysis of the data collected for the ORJIP project. This is likely, but not certain, to give an avoidance rate that is higher than the value recommended by NE for existing CRM calculation.

Adults foraging from the colony at FFC pSPA have been tracked by the RSPB as part of the FAME and STAR projects. While these data have been presented in summary form by the RSPB, they have not been used to estimate the magnitude of breeding season connectivity of this colony with existing and planned wind farms. Such analysis may be planned or underway by the RSPB, or could be conducted by other researchers if these data could be obtained. The habitat utilisation maps which could be developed would allow refinement of the extent of connectivity with existing wind farms (hence could reduce the number of collisions attributed to FFC pSPA) and would also permit identification of areas of higher and lower risk for future wind farm development. As tracking has been conducted over several years it should be possible to determine the extent of consistency in site selection between years.

7.3. Lesser black-backed gull

Breeding numbers in the UK increased by 29% from the national census in 1969-70 to 1985-88, and by a further 40% from 1985-88 to 1998-2002. Numbers seem to have remained approximately stable or declined from 2000 to 2015, but with strong variation in trend between colonies and little statistical confidence in trend estimates (http://jncc.defra.gov.uk/page-2886). With some evidence for increases in urban nesting numbers and some licenced culling of breeding birds, this species appears not to be a strong focus of conservation efforts.

Lesser black-backed gull numbers decline in UK waters in winter, so the main concern in relation to offshore wind farm impacts is likely to be in combination impacts of collision mortality for breeding birds from SPA colonies closest to developments. This will particularly apply to the Alde-Ore Estuary SPA population, the only SPA for breeding lesser black-backed gulls on the east coast of England. Stroud et al. (2016) estimated that about 38.5% of the GB population breeds within SPAs. The Alde-Ore Estuary SPA, the SPA for this species closest to many offshore wind farms in UK southern North Sea waters, held around 22,000 pairs in the early 1990s but numbers decreased to 6,000 pairs in 2003 (Stroud et al. 2016; although this was due primarily to changes in agriculture in the area removing food resources). Other SPA populations of concern are likely to be the Forth Islands population in east Scotland, and the six SPA populations around the Irish Sea.

The main option for further increasing lesser black-backed gull collision mortality headroom (i.e. measures which would be effective across most wind farms) would be to derive evidence based estimates of nocturnal flight activity. Data to refine nocturnal flight activity have already been collected by other research projects so this would be fairly straightforward to conduct once the data were obtained.

7.4. Great black-backed gull

Breeding numbers in the UK decreased by 7% from the national census in 1969-70 to 1985-88, and by a further 4% from 1985-88 to 1998-2002, and have continued to decrease by an estimated 11% from 1998-2002 to 2015 (http://jncc.defra.gov.uk/page-2888). Decreases in breeding numbers have been particularly large at some of the largest colonies in northern Britain, whereas numbers have increased in SW England (especially in the Isles of Scilly).

There is only one SPA for breeding great black-backed gulls in England, the Isles of Scilly SPA, while there are five in the far north of Scotland. Breeding numbers have declined in all five Scottish SPA colonies. Together the SPAs hold only about 17% of the GB breeding population (Stroud et al. 2016) and are distant from southern North Sea offshore wind farms so HRA is not generally an issue for this species in relation to UK offshore wind farms. There also appear to have been decreases in numbers coming to UK waters in winter from overseas (predominantly north Norway). With some licenced culling of breeding birds, this species appears not to be a strong focus of conservation efforts.

As a scavenging species, feeding extensively on fishery discards, especially in the nonbreeding period, this species may decrease in UK waters as the landings obligation results in further reduction in fishery discards. The main concern in relation to offshore wind farm developments is cumulative impacts of collision mortality at the EIA scale, which is most likely to affect the wintering population in the North Sea. An uncertain, but probably high proportion of these birds are from north Norwegian colonies. In addition, older wind farm assessments rarely considered this species, making it difficult to estimate robust cumulative collision totals.

The main options for further increasing great black-backed gull collision mortality headroom (i.e. measures which would be effective across most wind farms) would be to derive evidence based estimates of nocturnal flight activity and track nonbreeding season movements. This species has been little studied to date, so it would be necessary to conduct studies to derive these estimates. A geolocator logger study of breeding birds from colonies in UK, Norway and the Faeroes would provide data for both of these objectives.

7.5. Herring gull

This species increased enormously in breeding numbers in the UK from 1900 to about 1970, but has declined considerably at many colonies since 1970. Breeding numbers in the UK decreased by 48% from the national census in 1969-70 to 1985-88, and by a further 13% from 1985-88 to 1998-2002. Trends since 2000 are uncertain, and appear to vary among regions and colonies (http://jncc.defra.gov.uk/page-2887). Stroud et al. (2016) estimated that about 12.5% of the GB population breeds within SPAs. With evidence for continuing increases in urban nesting numbers and some licenced culling of breeding birds, this species appears not to be a strong focus of conservation efforts, although it has been red listed due to the large decline.

As a scavenging species using a wide range of foods including fishery discards, domestic and agricultural terrestrial wastes, the herring gull is likely to be adversely affected by forthcoming (further) reductions in fishery discards and dumping of food waste to land-fill. In the southern North Sea, collision risk of this species tends to be highest in winter, when numbers peak and include birds from north Norway as well as birds from UK colonies. The main concern is likely to be over cumulative collision mortality at the EIA scale. There may be some HRA concerns; herring gull is a feature of six SPAs on the east coast of Scotland, including Forth Islands SPA. It is not yet a designated feature of any SPAs for breeding seabirds in east England, but Stroud et al. (2016) indicate that it could qualify under SPA Guidelines selection stage 1.3 at the Alde-Ore Estuary and at Flamborough & Filey Coast, and at Stage 1.2 at Morecambe Bay.

The main option for further increasing herring gull collision mortality headroom (i.e. measures which would be effective across most wind farms) would be to derive evidence based estimates of nocturnal flight activity. There is some suitable data from past studies which could be used for this, although it may also be necessary to conduct further studies to improve the sample size.

8. Discussion

As the number of offshore wind farms has increased in the UK, collision mortality has become one of the main consenting risks. The number of wind turbine collisions is predicted using a model which combines seabird flight activity, species specific biometric estimates and turbine parameters. Uncertainty about the appropriate values for some of these parameters typically results in the use of precautionary estimates. As the number of wind farms has increased, the increase in cumulative mortality predictions has resulted in efforts to reduce the degree of precaution applied. The main collision model parameter to which this has been applied is the collision avoidance rate. This value is used to account for the expected turbine avoidance behaviour birds exhibit. This parameter is required by the model because predictions are derived from surveys conducted prior to wind farm construction when there are no objects for the birds to avoid. The avoidance rate has a large effect on the final collision predictions and thus change to this value has a big effect. It is also straightforward to apply avoidance rate revisions retrospectively to collision predictions made using lower rates, thereby enabling update of cumulative estimates. The BTO undertook a review of monitoring data at operational wind farms (Cook et al. 2014) which resulted in the avoidance rate for gannet and kittiwake being increased from 98% to 98.9% and for large gulls from 98% to 99.5%. These reduced the collision estimates for gannet and kittiwake to almost half of the previous totals and for the large gulls to a quarter of the previous totals. Even so, it is probable that for these species the avoidance rates remain precautionary, however it is likely that a considerable amount of additional monitoring will be required before any further revisions are possible.

Although the values for some other parameters in the collision model are probably precautionary, many of these relate to aspects of the species' biology (e.g. nocturnal activity estimates, flight heights, etc.) and will therefore require development of more accurate survey methods or long term studies before robust updates are feasible. The work presented here has focussed on the physical characteristics of the wind turbines. These are mostly known with certainty and therefore much less subject to debate (blade pitch is possibly an exception to this, as to date this has not been modelled in relation to wind speed). Consequently, updating the original collision predictions made on the basis of planned turbines, using parameters for the constructed wind farms does not require development of a supporting evidence base. The only reason that the revised collision estimates presented here (for operational wind farms and those under construction) may not be considered acceptable by SNCBs relates to the planning consents. These allow for wind farm construction based on worst case designs which are typically based on larger numbers of smaller turbines which generate higher collision risks. However, wind farm developments typically make use of larger turbines as this enables them to achieve consented generating capacities with fewer installed turbines. The problem is, that while the consent remains valid, the developer retains the option to construct a wind farm which corresponds to the upper limits allowed. Clearly, the likelihood of this actually occurring is very small, however there is a strong reluctance on the part of the SNCB's to accept this when assessing cumulative effects. This is the main barrier to acceptance of the revised cumulative collisions presented here.

In order to future proof the headroom calculations produced here, a spreadsheet format has been used. This comprises three primary tables:

- Wind farm data
- Species specific probability of collision (single rotor transit) calculations
- Collision recalculations

The species-specific probabilities of collision and collision recalculations use data contained in the wind farm data sheet. Therefore, updates to the wind farm values in this sheet will transfer across to the collision estimates.

The other aspect which determines collision mortality headroom is the tolerable limit of allowable mortality. No explicit limits have been defined by the SNCBs, therefore in lieu of such estimates we have used the last consented cumulative totals (or values accepted during project examination by NE) as an indication of the minimum threshold value. Consequently, the calculated headroom collision mortalities are also minimum estimates on the basis of current methods and data. As the designs for more recently consented projects become publicly available these can be incorporated into the calculations and will (almost certainly) reduce collisions further. Revisions to the collision modelling methods and parameter values used would also alter the predictions. While these are likely to reduce predicted collision rates, it is possible that monitoring studies may report collision rates higher than those predicted, which could lead to upward revision of cumulative mortality. Furthermore, the thresholds are also subject to change due to change in the size and status of the relevant populations. Wind farm collisions can be assessed against a range of population scales, from biogeographic to individual SPA breeding colonies. As the population scale reduces in size across this range, so the relative impact increases (although this is partially offset by the smaller number of wind farms contributing to the total mortality). It is therefore not surprising that the main concern for wind farm impact assessments is usually at the level of individual SPA populations. This is compounded by the requirement to assess impacts against SPA populations in isolation, despite the fact that seabird breeding colonies are connected through immigration and emigration. Changes in the status of individual SPA populations is therefore likely to large effects on acceptable mortality levels.

Of the species most at risk of collision, which are also well represented in the UK SPA suite, kittiwake is likely to be the species for which there is most concern, largely due to widespread declines at most colonies. Gannet is also often cited as a species of concern, although this is largely a reflection of the high proportion of this species which breeds at British colonies and the consequent responsibility to safeguard the species. The population is still growing (in contrast to most other seabird species) and there is increasing evidence that gannets have very high wind farm avoidance (probably higher than the current 98.9% avoidance rate). However, set against this is the fact that the headroom for gannet is less than that for kittiwake. Thus, it seems likely that one or other of these species will be the subject of primary concern for future North Sea wind farm developments, depending on the location.

Few Irish Sea wind farm assessments have presented collision estimates for gannet and kittiwake, which is presumably a reflection of low densities of these species (it is also notable that there are relatively few SPAs for these species found around the Irish Sea). Therefore, there is scope for further wind farm expansion in relation to impacts on these species. However, lesser black-backed gull and herring gull are present in higher numbers within this region and are features of several SPAs. Consequently these species have been the focus of wind farm assessments. In addition, large numbers of common scoters and red-throated divers over-winter in the Irish Sea and are subject to displacement effects. Therefore, while collision risks for certain species may not be limiting, other species and other impacts also need to be considered.

The south coast has comparatively few seabird SPAs and the SeaMast data suggest quite low densities. The main collision risk posed by wind farms in this region is likely to be to birds on migration, which would be expected to be a smaller risk than that for breeding birds at wind farms within foraging range.

The other species most commonly considered at widespread risk of collisions are the large gulls assessed in this study. While these species are features of a few SPAs, they are generally present in smaller numbers, have relatively short foraging ranges and have low predicted numbers of breeding season collisions. It is generally at wider population scales that concerns have been raised. Nonbreeding season collision predictions tend to be high for great black-backed and herring gull due to influxes of birds into the North Sea from colonies further north (e.g. Norway). However, while the collision predictions increase, so too do the population sizes, which has tended to offset concerns. Lesser black-backed gulls migrate away from the North Sea in winter, so for this species the focus is more on breeding season collisions, but these are localised and generally quite low. The very high large gull avoidance rates have also greatly reduced predicted impacts and concerns and these species are unlikely to be significant consenting risks for future wind farm development.

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Appendix 1: SPA Collision risk species foraging range overlaps with existing offshore wind farms

Lesser black-backed gull

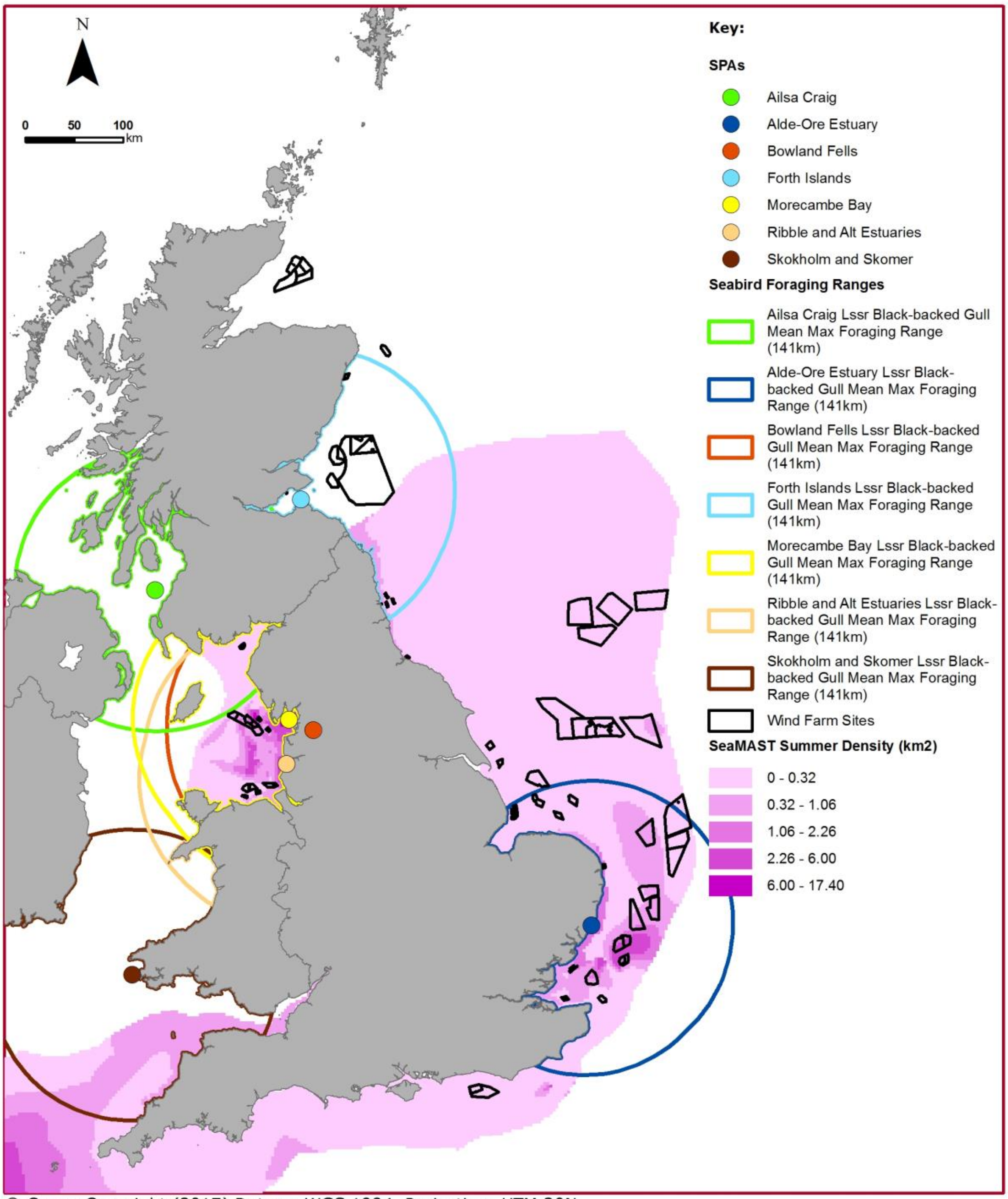

Figure A1.1. Density of lesser black-backed gulls in summer (SeaMAST output) overlaid with mean maximum foraging range from key SPAs with potential for connectivity to wind farms in English waters.

Table A1.1. Annual in-combination collision mortality for lesser black-backed gulls from Ribble and Alt Estuary SPA, Morecambe Bay SPA, Bowland Fells SPA and Alde Ore Estuary SPA at UK offshore wind farms. Headroom is the difference between the original and updated collision estimates.

Species	**Ribble and Alt Estuary SPA**			**Morecambe Bay SPA**			**Bowland Fells SPA**			**Alde Ore Estuary SPA**		
	Original	Updated	**Headroom**	Original	Updated	**Headroom**	Original	Updated	**Headroom**	Original	Updated	**Headroom**
Lesser black-backed gull	53	32	21	76	58	18	6	5	1	65	39	**26**

Herring gull

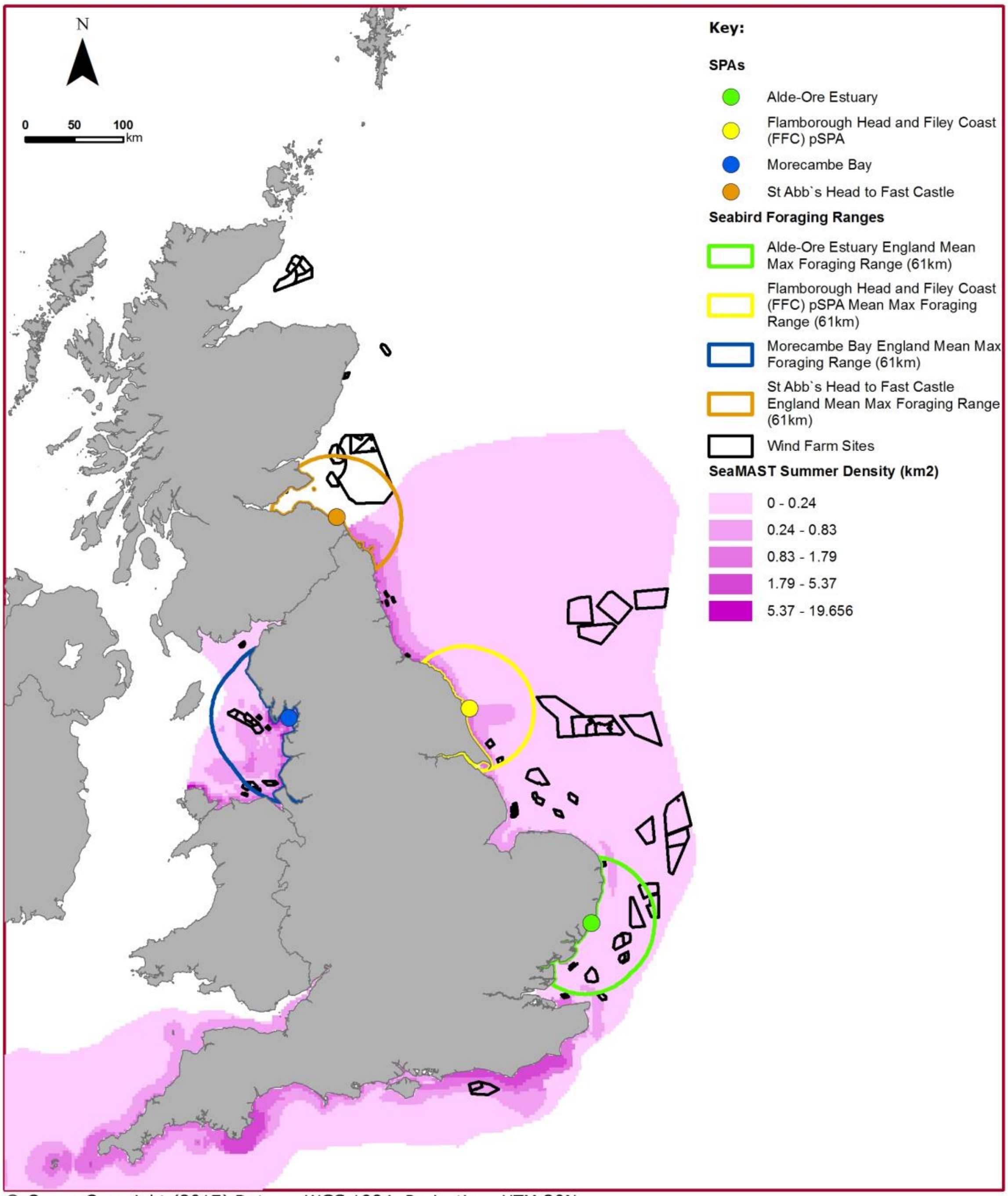

Figure A1.2. Density of herring gulls in summer (SeaMAST output) overlaid with mean maximum foraging range from key SPAs with potential for connectivity to wind farms in English waters.

Table A1.2. Annual in-combination collision mortality for herring gulls from Morecambe Bay SPA and Alde Ore Estuary SPA at UK offshore wind farms. Headroom is the difference between the original and updated collision estimates.

Species	Morecambe Bay SPA			Alde Ore Estuary SPA		
	Original	Updated	**Headroom**	Original	Updated	**Headroom**
Herring gull	7	3	4	8	3	**5**

Sandwich tern

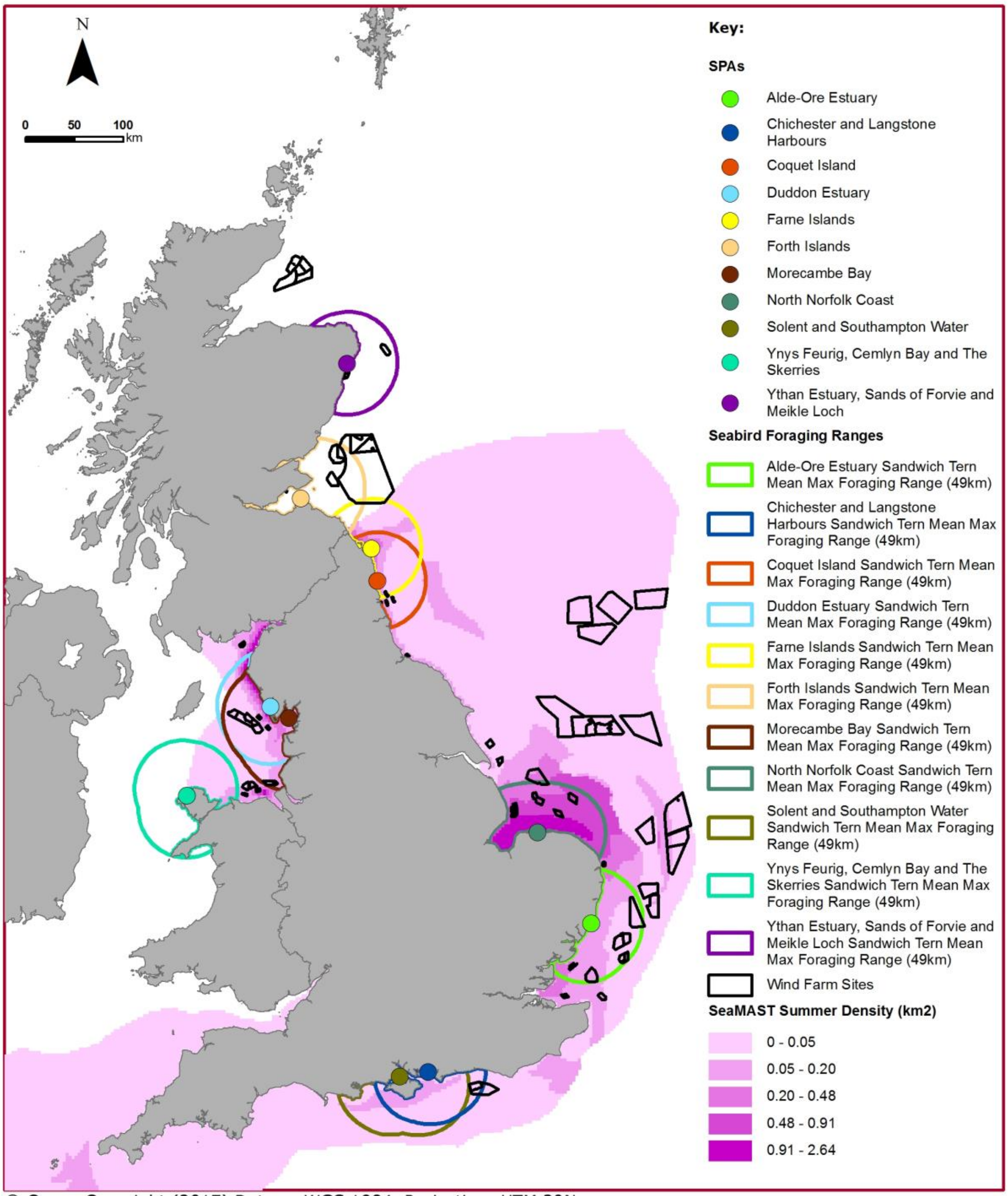

Figure A1.3. Density of Sandwich terns in summer (SeaMAST output) overlaid with mean maximum foraging range from key SPAs with potential for connectivity to wind farms in English waters.

Table A1.3. Annual in-combination collision mortality for Sandwich tern from North Norfolk Coast SPA at UK offshore wind farms. Headroom is the difference between the original and updated collision estimates.

Species	North Norfolk Coast SPA		
	Original	Updated	**Headroom**
Sandwich tern	94	58	36

Appendix 2: Apportioning of the Flamborough and Filey Coast pSPA gannet population among North Sea offshore wind farms

(https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010056/EN010056-000556-5.4%20(3)%20Information%20to%20Inform%20HRA%20Appendix%203.pdf)

East Anglia
Offshore Wind

VATTENFALL

SCOTTISHPOWER
RENEWABLES

East Anglia THREE

Information for Habitats Regulations Assessment

Appendix 3 Apportioning of the Flamborough Head and Filey Coast pSPA Gannet Population among North Sea Offshore Windfarms

Document Reference – 5.4 (3)

Author – MacArthur Green
East Anglia THREE Limited
Date – November 2015

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1 INTRODUCTION

1. This report provides an estimation of the percentage of the Flamborough and Filey Coast (FFC) pSPA gannet population at risk of collision effects in North Sea offshore windfarms. This report follows the methods used for previous assessments (MacArthur Green 2014, 2015), but presents updated calculations using breeding colony estimates presented in Murray et al. (2015).

2 ESTIMATION OF UK NORTH SEA & CHANNEL WATERS GANNET BDMPS AND FFC PSPA PROPORTION

2. Due to variations in the timing of migration among individuals both within and between colonies and also between different age classes there is considerable overlap in the gannet seasons for the UK (Furness 2015; *Table 2.1*).

Table 2.1. Gannet seasons in UK waters from Furness (2015).

Season	J	F	M	A	M	J	J	A	S	O	N	D
Non-breeding	■	■								■	■	■
Spring migration (UK waters)	■	■	■									■
UK Breeding season (full)			■	■	■	■	■	■	■			
UK Breeding season (core)				■	■	■	■	■				
Autumn migration (UK waters)									■	■	■	

3. For this assessment the following descriptions of the seasons are considered to be appropriate:

- Spring migration (December – March)
- UK breeding (April – August)
- Autumn migration (September – November).

2.1 Migration periods

4. During migration periods gannets from different breeding colonies mix together to varying extents but show some segregation in wintering latitude and migration routes (Fort et al. 2012). No birds tracked from the Bass Rock population in 2008-09 spent the winter in the North Sea (Garthe et al. 2012), and passage routes have shown that approximately 63% fly south through the North Sea and English Channel in autumn, while 27% follow the same route north in spring (WWT 2012). Other studies of tracked birds (summarised in WWT 2012) have been used here to generate estimates of movements through the North Sea. On the basis of these data, and observations made at coastal observatories (summarised in Furness 2015)), the proportions of the FFC population expected to be present in the UK North Sea and English Channel during the autumn and spring passage periods are estimated in *Table 2.2*.

5. The estimates in *Table 2.2* make use of flight direction data derived from the detailed tagging studies which have been conducted on birds from the FFC pSPA and Bass Rock colonies. This approach is slightly different from that presented in Furness (2015), where proportions were estimated without specific reference to these

tagging data. The use of these data here also results in a slightly different approach to how migration of different age classes was calculated.

6. Only adults had tags fitted, therefore we only have proportional movement estimates for adults. We have therefore assumed that the proportions of immature birds which travel north and south from their colonies are the same as for observed adults. This differs from the age related differences reported in Furness (2015) and leads to some further small differences between this assessment and the numbers reported in Furness (2015). For example, in the latter a small number of immature birds from the west coast colonies at Grassholm and Ailsa Craig have been predicted to have connectivity with the North Sea. In this assessment, since we have not estimated immature movement rates separately from adults, we have taken a slightly more precautionary approach; only colonies with predicted adult connectivity to the North Sea have been included.

7. Furthermore, there are some minor differences in how colonies have been presented in *Table 2.2* and in Furness (2015). These differences reflect colony status (i.e. whether or not they are included in the relevant SPA citation) and the potential for connectivity with the North Sea, but the actual population sizes used are the same in both this report and Furness (2015). For example, *Table 2.2* includes the named colonies (Foula, West Westray and Troup Head) as these are important components of the North Sea population (although gannet are not currently included on the relevant SPA citations), while in Furness (2015) these colonies are included within the category of 'UK North Sea non-SPA colonies' (Furness (2015); Table 7.1 and Appendix Table 14). These sites have been included in the current assessment on the grounds of their connectivity with the North Sea.

8. The population sizes used here include all age classes, estimated by dividing the breeding population by the adult proportion (0.55; Furness 2015).

2.2 Breeding season

9. During the core breeding season gannets observed in windfarm sites in the North Sea are assumed to originate from one of two candidate colonies: FFC and the Bass Rock (BR). For windfarm sites located to the south of FFC and within the maximum foraging range (590km Thaxter et al. 2012) it has been assumed that all breeding season observations are birds from FFC. Wakefield et al. (2013) present evidence indicating there is relatively little overlap in foraging areas between breeding colonies. However, it remains possible that windfarm sites located within the zone where birds from the two colonies may forage have been assigned equally between the two populations.

Table 2.2. Total population estimates for gannet breeding populations with connectivity to the North Sea on passage (using adult percentage of 55% from Furness (2015); see text for description of region extent). Note that AON are the values presented in Murray et al. (2015) where provided.

Breeding colony (year of count)	AON	No. adults	All ages	Autumn migration through N Sea & English Channel				Spring migration through N Sea and English Channel			
				Fly south		Fly north		Fly north		Fly south	
				Prop.	No.	Prop.	No.	Prop.	No.	Prop.	No.
Iceland (1999)	37216	74432	134722	0.42	56583	0	0	0.1	13472	0	0
Norway (2010)	6000	12000	21720	0.5	10860	0	0	0.2	4344	0	0
Faeroes (2012)	2500	5000	9050	0.42	3801	0	0	0.2	1810	0	0
Hermaness (2008)	25580	51160	92600	0.5	46300	0	0	0.5	46300	0	0
Noss (2008)	11786	23572	42665	0.5	21333	0	0	0.5	21333	0	0
Foula (2007)	1226	2452	4438	0.5	2219	0	0	0.5	2219	0	0
Fair Isle (2013)	3591	7182	12999	0.5	6500	0	0	0.5	6500	0	0
West Westray (2012)	751	1502	2719	0.5	1359	0	0	0.5	1359	0	0
Sule Skerry & Sule Stack (2004)	6420	12840	23240	0.1	2324	0	0	0	0	0	0
North Rona and Sula Sgeir (2004)	11230	22460	40653	0.1	4065	0	0	0	0	0	0
St. Kilda (2004)	60290	120580	218250	0.1	21825	0	0	0	0	0	0
Troup Head (2010)	6456	12912	23371	0.63	14724	0.37	8647	0.27	6310	0.73	17061
Bass Rock (2009)	75259	150518	272438	0.63	171636	0.37	100802	0.27	73558	0.73	198879
Flamborough Head and Filey Coast (2012)	11061	22122	40041	0.75	30031	0.25	10010	0.5	20020	0.5	20020
Helgoland (2004)	656	1312	2375	1	2375	0	0	1	2375	0	0
Total	260022	520044	941280		395934		119459		199600		235960

2.3 Estimation of the percentage of the FFC pSPA gannet population at risk of collision effects in North Sea offshore windfarms

10. Using the estimated movement patterns of gannets through UK waters, seasonal definitions and regional definitions the percentage of gannets within North Sea offshore wind farms originating from the FFC pSPA colony is provided in *Table 2.3*. At wind farms within foraging range of the FFC pSPA (but beyond foraging range of all other colonies) all the gannets seen during the breeding season have been assumed to originate from FFC. For sites which are located within potential range of both FFC and Bass Rock it has been assumed that birds are equally likely to originate from either colony (hence 50% are attributed to FFC). This was based on consideration of the comparative ranges and site locations which indicate that while these sites are not equidistant to both colonies, they do fall within the region of overlapping foraging ranges.

11. The breeding season values reflect estimates for all age classes combined. To estimate the number of breeding adults from the FFC pSPA at risk, the breeding season percentages provided in *Table 2.3* need to be adjusted by the estimated proportion of this age class present (a value of 0.55 has been assumed for this metric, Furness 2015).

12. During migration the number of British SPA birds travelling north and south through the North Sea has been summed using the population estimates and proportions in Table 2. The British total was then added to the numbers estimated to pass through the North Sea which originate from Norwegian, Icelandic and Faeroese colonies. Having estimated the total flux of gannets on passage, the FFC pSPA percentage of this was then calculated for each windfarm in *Table 2.3* on the basis of its location relative to the colony locations.

13. The following examples illustrate the method used to calculate the number of birds from the FFC pSPA which pass through OWFs in each region of the North Sea and also the number estimated to remain in the North Sea over the winter. The calculations use the population estimates and the proportions predicted to pass through the North Sea in *Table 2.2*.

14. During autumn migration, the proportion of adult birds from FFC passing through OWFs in the Moray Firth has been calculated as:

- The no. of adults from FFC (11,061 x 2) = 22,122 multiplied by the proportion of this population estimated to migrate north through the Moray Firth (0.25) =

5,530; this is then divided by the total number of all age birds estimated to pass through the Moray Firth on migration (derived below).

- The total number on passage through the Moray Firth includes birds travelling both south (from farther north colonies) and north (from farther south colonies):
    - The number of all age birds from non-GB colonies (165,492) multiplied by the proportion estimated to pass through the Moray Firth (0.5 for Norway, 0.42 for Iceland and Faeroes) = 71,244, plus
    - The number of all age birds from UK colonies north of the Moray Firth which are predicted to have connectivity to the North Sea (Noss, Hermaness, Foula, Fair Isle, West Westray, Sule Skerry and Sule Stack, North Rona and Sula Sgeir, St. Kilda; 437,564) multiplied by the proportion estimated to pass southwards through the Moray Firth (0.5 & 0.1) = 105,925, plus
    - The number of all age birds from North Sea colonies south of the Moray Firth (Troup Head, Bass Rock, FFC; 335,849) multiplied by the proportion estimated to pass north through the Moray Firth (0.37 for Troup and Bass Rock, 0.25 for FFC) = 119,459.
- = 5530 / (71244 + 105925 + 119459)
- = 0.019

15. Therefore, 1.9% of the gannets which pass through Moray Firth OWFs on autumn passage are estimated to be breeding adults from the FFC pSPA population.

16. This process was conducted for all North Sea OWFs under consideration, split into the following geographical categories:

- Moray Firth (Beatrice and MORL)
- Aberdeen (EOWDC)
- Firths of Forth and Tay (Inch Cape, Neart na Goithe, SeaGreen)
- NE England (Blyth, Dogger Bank CB A&B, Teesside A&B and Teesside C&D, and Teesside)
- SE England (Dudgeon, EA1, Hornsea Projects One and Two, Humber Gateway, Galloper, Gtr Gabbard, Lincs, London Array, Race Bank, Sheringham, Thanet, Triton Knoll, Westermost Rough)

- S England (Rampion and Navitus Bay)

17. For each geographical division the FFC passage number was divided by the summed contribution from colonies to the north and south on the basis of the proportions travelling in each direction (N or S). The same method was used for calculating FFC proportions passing through the OWFS during spring migration.

Table 2.3. Percentage of gannets in offshore wind farms during the breeding season (BS), autumn migration (Aut.), and spring migration (Spr.) periods which are estimated to originate from the FFC pSPA population.

Project	UK Round	Status	Period		
			BS	Aut.	Spr.
Beatrice	Scottish	Consent Authorised	0	1.9	3.3
Blyth Demonstration Site	-	Consent Authorised	0	1.5	5.6
Dogger Bank Creyke Beck A & B	3	Consent Authorised	50	1.5	5.6
Dogger Bank Teesside A & B	3	Consent Authorised	50	1.5	5.6
Dogger Bank Teesside C & D	3	Concept / Early Planning	50	1.5	5.6
Dudgeon	2	Consent Authorised	100	4.2	5.6
East Anglia ONE	3	Consent Authorised	100	4.2	5.6
East Anglia THREE	3	Consent Application Submitted	100	4.2	5.6
European Offshore Wind Development Centre		Consent Authorised	0	1.8	3.4
Firth of Forth Alpha and Bravo (Seagreen)	Scottish	Consent Authorised	0	1.8	3.4
Galloper	2 – extn.	Consent Authorised	0	4.2	5.6
Greater Gabbard	2	Fully Commissioned	0	4.2	5.6
Hornsea Project One	3	Consent Authorised	100	4.2	5.6
Hornsea Project Two	3	Consent Application Submitted	100	4.2	5.6
Humber Gateway	2	Consent Authorised	100	4.2	5.6
Inch Cape	Scottish	Consent Authorised	0	1.8	3.4
Lincs	2	Partial Generation / Construction	100	4.2	5.6
London Array	2	Fully Commissioned	0	4.2	5.6
Moray	Scottish	Consent Authorised	0	1.9	3.3
Navitus Bay	3	Consent Application Submitted	0	4.2	5.6
Neart na Gaoithe	Scottish	Consent Authorised	0	1.8	3.4
Race Bank	2	Consent Authorised	100	4.2	5.6
Rampion	3	Consent Authorised	0	4.2	5.6
Sheringham Shoal	2	Fully Commissioned	100	1.9	3.3
Teesside	1	Fully Commissioned	50	1.5	5.6
Thanet	2	Fully Commissioned	0	1.5	5.6
Triton Knoll	2	Consent Authorised	100	1.5	5.6
Westermost Rough	2	Partial Generation / Construction	100	1.5	5.6

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Wakefield, E.D., Bodey, T.W., Bearhop, S., Blackburn, J., Colhoun, K., Davies, R., Dwyer, R.F., Green, J.A. Gremillet, D., Jackson, A.L., Jessopp, M.J., Kane, A., Langston, R.H.W., Lescroel, A., Murray, S., Le Nuz, M., Patrick, S.C., Peron, C., Soanes, L.M., Wanless, S., Votier, S.C. and Hamer, K.C. (2013). Space partitioning without territoriality in gannets. *Science* 341: 68-70.

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Appendix 3: Apportioning of the Flamborough and Filey Coast pSPA kittiwake population among North Sea offshore wind farms

(https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010056/EN010056-000557-5.4%20(4)%20Information%20to%20Inform%20HRA%20Appendix%204.pdf)

East Anglia Offshore Wind

VATTENFALL

SCOTTISHPOWER RENEWABLES

East Anglia THREE

Information for Habitats Regulations Assessment

Appendix 4 : Apportioning of the Flamborough Head and Filey Coast pSPA Kittiwake Population among North Sea Offshore Wind Farms

Document Reference – 5.4 (4)

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Date – November 2015

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1 INTRODUCTION

1. This report provides an estimation of the percentage of the Flamborough and Filey Coast (FFC) pSPA kittiwake population at risk of collision effects in North Sea offshore wind farms. This report follows the methods used for previous assessments (MacArthur Green 2014, 2015).

2 ESTIMATION OF UK NORTH SEA & CHANNEL WATERS KITTIWAKE BDMPS AND FFC PSPA PROPORTION

2. Due to variations in the timing of migration among individuals both within and between colonies and also between different age classes there is considerable overlap in the kittiwake seasons for the UK (Furness 2015; *Table 2.1*).

Table 2.1. Kittiwake seasons in UK waters from Furness (2015).

Season	J	F	M	A	M	J	J	A	S	O	N	D
Non-breeding (core)	■											■
Non-breeding (full)	■	■							■	■	■	■
Spring migration (UK waters)	■	■	■	■								
UK Breeding season (full)			■	■	■	■	■	■				
UK Breeding season (core)					■	■	■					
Autumn migration (UK waters)								■	■	■	■	■

3. Despite the brief core winter period with little migration, the BDMPS report concluded that it was more appropriate to define two nonbreeding seasons for kittiwake; autumn migration and spring migration, rather than separating a third ‘winter’ season (Furness 2015). For this assessment the following descriptions of the seasons are considered to be appropriate:

- Spring migration (January – April)
- UK breeding (May – July)
- Autumn migration (august - December).

2.1 Migration periods

4. During the migration periods kittiwakes from different breeding colonies mix together to varying extents. *Table 2.2* presents the population size for all British kittiwake SPA colonies plus those populations further north which are thought to contribute to UK North Sea passage and wintering populations. It should be noted that only approximately half of the British kittiwake population breeds at SPAs for this species. The proportions of each of the populations considered which is predicted to pass through the North Sea have been derived from Furness (2015).

5. The most recent estimates for the two large SPA colonies in Caithness (North Caithness Cliffs SPA and East Caithness Cliffs SPA) date from 1999 and 2000. Therefore, it has been suggested that since these counts were undertaken these populations may have declined by a similar margin (e.g. up to 50%) as reported for other colonies in northern Scotland (Furness 2015). However, there is evidence to

suggest that this is not likely to have been the case. Site condition monitoring (SCM) of the East Caithness Cliffs SPA (Swann 2012) reported that the kittiwake population within monitoring plots had in fact increased by 65% between 1999 and 2005, in contrast to the declines reported elsewhere (Mavor et al. 2005). It is thus far from certain that the East Caithness Cliffs SPA population has declined. There is no published SCM for the North Caithness Cliffs SPA, however this site is close to East Caithness and therefore it is reasonable to assume similar trends at both. Furthermore, since it makes a comparatively small contribution to the total apportioning calculations it was considered more appropriate to retain the reported census from 2000 rather than make assumptions about an unknown trend.

Table 2.2. Total population estimates for major breeding populations (using adult percentage of 53.2 from Furness 2015) and proportion and number estimated to pass through or remain in, UK North Sea waters during migration.

Breeding colony (year of count)	AON	No. adults	No. immatures	Proportion and number in UK North Sea waters during autumn migration				Proportion and number in UK North Sea waters during spring migration			
				Adults		Immatures		Adults		Immatures	
				Prop.	No.	Prop.	No.	Prop.	No.	Prop.	No.
Russia (c. 2000)	140000	280000	246400	0.1	28000	0.1	24640	0.05	14000	0.07	17248
Norway (c. 2000)	700000	1400000	1232000	0.1	140000	0.1	123200	0.05	70000	0.07	86240
Faeroes (2012)	200000	400000	352000	0.1	40000	0.1	35200	0.05	20000	0.07	24640
Germany (2010)	6000	12000	10560	0.1	1200	0.1	1056	0.15	1800	0.25	2640
France (2010)	4000	8000	7040	0.05	400	0.05	352	0.05	400	0.1	704
Ireland (2000)	20000	40000	35200	0.05	2000	0.05	1760	0.01	400	0.01	352
Hermaness, Saxavord & Valla (2009)	391	782	688	0.6	469	0.40	275	0.6	469	0.30	206
Foula (2013)	327	654	576	0.6	392	0.40	230	0.6	392	0.30	173
Noss (2010)	507	1014	892	0.6	608	0.40	357	0.6	608	0.30	268
Sumburgh Head (2013)	210	420	370	0.6	252	0.40	148	0.6	252	0.30	111
Fair Isle (2013)	771	1542	1357	0.6	925	0.40	543	0.6	925	0.30	407
West Westray (2007)	12055	24110	21217	0.6	14466	0.40	8487	0.6	14466	0.30	6365
Calf of Eday (2006)	747	1494	1315	0.6	896	0.40	526	0.6	896	0.30	394
Marwick Head (2013)	526	1052	926	0.6	631	0.40	370	0.6	631	0.30	278
Rousay (2009)	1764	3528	3105	0.6	2117	0.40	1242	0.6	2117	0.30	931
Copinsay (2012)	666	1332	1172	0.6	799	0.40	469	0.6	799	0.30	352
Hoy (2007)	397	794	699	0.6	476	0.40	279	0.6	476	0.30	210
North Caithness Cliffs (2000)	10150	20300	17864	0.6	12180	0.40	7146	0.6	12180	0.30	5359
East Caithness Cliffs (1999)	40410	80820	71122	0.6	48492	0.40	28449	0.6	48492	0.30	21336
Troup, Pennan and Lion's Heads (2007)	14896	29792	26217	0.6	17875	0.40	10487	0.6	17875	0.30	7865
Buchan Ness to Collieston Coast (2007)	12542	25084	22074	0.6	15050	0.40	8830	0.6	15050	0.30	6622
Fowlsheugh (2012)	9337	18674	16433	0.6	11204	0.40	6573	0.6	11204	0.30	4930
Forth Islands (2013)	3100	6200	5456	0.6	3720	0.40	2182	0.6	3720	0.30	1637
St Abb's Head to Fast Castle (2013)	3403	6806	5989	0.6	4084	0.40	2396	0.6	4084	0.30	1797
Farne Islands (2013)	3443	6886	6060	0.6	4132	0.40	2424	0.6	4132	0.30	1818

Flamborough and Filey Coast (2008)	37617	75234	66206	0.6	45140	0.40	26482	0.6	45140	0.30	19862
UK North Sea non-SPA colonies (2000)	70000	140000	123200	0.6	84000	0.40	49280	0.6	84000	0.30	36960
Cape Wrath (2000)	10344	20688	18205	0.01	207	0.05	910	0.01	207	0.02	364
North Rona and Sula Sgeir (2012)	1253	2506	2205	0.01	25	0.05	110	0.01	25	0.02	44
Handa (2013)	1872	3744	3295	0.01	37	0.05	165	0.01	37	0.02	66
St Kilda (2008)	957	1914	1684	0.01	19	0.05	84	0.01	19	0.02	34
Flannan Isles (1998)	1392	2784	2450	0.01	28	0.05	122	0.01	28	0.02	49
Shiant Isles (2008)	549	1098	966	0.01	11	0.05	48	0.01	11	0.02	19
Canna and Sanday (2013)	820	1640	1443	0.01	16	0.05	72	0.01	16	0.02	29
Rum (2000)	788	1576	1387	0.01	16	0.05	69	0.01	16	0.02	28
Mingulay and Berneray (2009)	2228	4456	3921	0.01	45	0.05	196	0.01	45	0.02	78
North Colonsay & Western Cliffs (2000)	5563	11126	9791	0.01	111	0.05	490	0.01	111	0.02	196
Ailsa Craig (2013)	489	978	861	0.01	10	0.05	43	0.01	10	0.02	17
Rathlin Island (2011)	7922	15844	13943	0.01	158	0.05	697	0.01	158	0.02	279
Skomer and Skokholm (2013)	1045	2090	1839	0.01	21	0.05	92	0.01	21	0.02	37
UK western non-SPA colonies (2000)	30000	60000	52800	0.01	600	0.05	2640	0.01	600	0.02	1056
Total	1365384	2730768	2403075		489096		353981		384096		255646

2.2 Breeding season

6. The mean maximum foraging range estimate for kittiwake is 60 km and the maximum range is estimated to be 120km (Thaxter et al. 2012). Since the East Anglia THREE site is 257km from the pSPA it is concluded there is no probability of any breeding adults from FFC being present on the wind farms during the breeding season. Birds recorded at this time are therefore assumed to be failed or non-breeders (including immature birds).

7. Determination of the colonies to which these non-breeding birds are associated can only be undertaken on the basis of assumptions about their movements. If it is assumed that these birds are derived equally from all North Sea colonies ('Hermaness' to 'UK North Sea non-SPA colonies', listed in *Table 2.2*) and that these birds mix uniformly throughout the North Sea this gives a percentage attributable to the FFC pSPA of 16.8% (75234 / 446518); note that although this percentage has been calculated using AON values, under the assumption of equivalent SPA contributions they apply equally to other age classes). However, these totals do not include immature birds from breeding populations farther north (Norwegian Sea and Barents Sea): immature birds move away from their natal colonies after fledging and then spend several years moving gradually closer to their natal breeding colonies (Wernham et al. 2002, Coulson 2011). Thus, immature birds from north Atlantic populations are likely to be present in the North Sea as a component of the overall population. Therefore, the value of 19.3%, attributable to the FFC pSPA population, derived solely on the basis of UK breeding populations, can be considered highly likely to be an overestimate. Given the large size of the Norwegian, Faeroes and Barents Sea kittiwake populations, including immature birds from these locations in the same manner as above (i.e. based on the equivalence of AON and immature birds) generates a revised percentage of 3.3% for FFC (75234 / (446518 + 1830400)). This is likely to overestimate the presence of immature north Atlantic birds in the North Sea, but does provide lower limits to bracket the range of likely percentages: 3.3% – 16.8%.

3. Estimation of the percentage of the FFC pSPA kittiwake population at risk of collision effects in UK North Sea waters offshore wind farms

8. On the basis of the estimated distribution of kittiwakes through UK waters, seasonal definitions and regional definitions (MacArthur Green 2014), the percentage of kittiwakes within named North Sea offshore wind farms which originate from the FFC pSPA colony has been calculated (*Table 2.3*). For wind farms located within 60 km of the FFC pSPA colony (Westermost Rough and Humber Gateway) it has been assumed that between 16.8 and 100% of the birds seen on site during the breeding

season originate from this population. For all others the breeding season percentage was assumed to be the precautionary North Sea breeding season value calculated above (19.3%).

9. During migration the number of British SPA birds travelling through UK North Sea waters has been summed using the population estimates and proportions in *Table 2.2*. This British total was then added to the numbers estimated to pass through UK North Sea waters which originate from Russian and Norwegian colonies. Having estimated the total flux of kittiwakes on passage, the FFC pSPA percentage of this was then calculated for each wind farm in *Table 2.3*.

10. The following examples illustrate the method used to calculate the number of birds from the FFC pSPA which pass through OWFs in each region of UK North Sea waters. The calculations use the population estimates and the proportions predicted to pass through the North Sea listed in *Table 2.2*.

11. On the assumption that during migration kittiwakes within UK North Sea waters originate from all contributing colonies in proportion to colony size, the migration period percentage estimated to originate from the FFC pSPA population on the wind farms listed in *Table 2.3* was calculated as the colony's proportion of the total (adults plus immatures).

12. Thus, the FFC proportion in UK North Sea waters present during autumn migration was estimated to be:

$$(75234 * 0.6) / (480812 + 349121) = 0.054$$

13. A similar calculation was conducted for the FFC spring migration period which gave an estimate of 0.072.

Table 2.3. Percentage of breeding adult kittiwakes in offshore wind farms during the breeding season (BS), autumn migration (Aut.) and spring migration (Spr.) periods which are estimated to originate from the FFC pSPA population.

<table>
<tr><th rowspan="2">Project</th><th rowspan="2">UK Round</th><th rowspan="2">Status</th><th colspan="3">Period</th></tr>
<tr><th>BS</th><th>Aut.</th><th>Spr.</th></tr>
<tr><td>Beatrice</td><td>Scottish</td><td>Consent Authorised</td><td rowspan="14">16.8</td><td rowspan="28">5.4</td><td rowspan="28">7.2</td></tr>
<tr><td>Blyth Demonstration Site</td><td>-</td><td>Consent Authorised</td></tr>
<tr><td>Dogger Bank Creyke Beck A & B</td><td>3</td><td>Consent Authorised</td></tr>
<tr><td>Dogger Bank Teesside A & B</td><td>3</td><td>Consent Authorised</td></tr>
<tr><td>Dogger Bank Teesside C & D</td><td>3</td><td>Concept / Early Planning</td></tr>
<tr><td>Dudgeon</td><td>2</td><td>Consent Authorised</td></tr>
<tr><td>East Anglia ONE</td><td>3</td><td>Consent Authorised</td></tr>
<tr><td>East Anglia THREE</td><td>3</td><td>Consent Application Submitted</td></tr>
<tr><td>European Offshore Wind Development Centre</td><td></td><td>Consent Authorised</td></tr>
<tr><td>Firth of Forth Alpha and Bravo (Seagreen)</td><td>Scottish</td><td>Consent Authorised</td></tr>
<tr><td>Galloper</td><td>2 – extn.</td><td>Consent Authorised</td></tr>
<tr><td>Greater Gabbard</td><td>2</td><td>Fully Commissioned</td></tr>
<tr><td>Hornsea Project One</td><td>3</td><td>Consent Authorised</td></tr>
<tr><td>Hornsea Project Two</td><td>3</td><td>Consent Application Submitted</td></tr>
<tr><td>Humber Gateway</td><td>2</td><td>Consent Authorised</td><td>16.8 - 100</td></tr>
<tr><td>Inch Cape</td><td>Scottish</td><td>Consent Authorised</td><td rowspan="12">16.8</td></tr>
<tr><td>Lincs</td><td>2</td><td>Partial Generation / Construction</td></tr>
<tr><td>London Array</td><td>2</td><td>Fully Commissioned</td></tr>
<tr><td>Moray</td><td>Scottish</td><td>Consent Authorised</td></tr>
<tr><td>Navitus Bay</td><td>3</td><td>Consent Application Submitted</td></tr>
<tr><td>Neart na Gaoithe</td><td>Scottish</td><td>Consent Authorised</td></tr>
<tr><td>Race Bank</td><td>2</td><td>Consent Authorised</td></tr>
<tr><td>Rampion</td><td>3</td><td>Consent Authorised</td></tr>
<tr><td>Sheringham Shoal</td><td>2</td><td>Fully Commissioned</td></tr>
<tr><td>Teesside</td><td>1</td><td>Fully Commissioned</td></tr>
<tr><td>Thanet</td><td>2</td><td>Fully Commissioned</td></tr>
<tr><td>Triton Knoll</td><td>2</td><td>Consent Authorised</td></tr>
<tr><td>Westermost Rough</td><td>2</td><td>Partial Generation / Construction</td><td>16.8 - 100</td></tr>
</table>

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APPENDIX 1: FACTORS AFFECTING SEABIRD FORAGING RANGES FROM BREEDING COLONIES

14. Foraging trip durations and maximum foraging ranges of many species are longer when prey abundance is reduced (Hamer et al. 1993, Lewis et al. 2006, Riou et al. 2011, Thaxter et al. 2012, Wade et al. 2014). In addition to that effect, they also tend to increase as a function of colony size, presumably due to intra-specific competition for prey resources at sea (Lewis et al. 2001, Forero et al. 2002, Ainley et al. 2003, Wakefield et al. 2013). Davies et al. (2013) showed that gannet foraging trips increased in duration and distance with colony size, but were considerably higher in a year when food abundance was thought to be lower than they were in a year of high food abundance. Furthermore, the slope of the relationship between colony size and foraging range was much higher in the year of lower food availability, suggesting a much stronger density-dependent effect when food resources are reduced. These results support the idea that foraging range relates to density-dependent competition, with larger ranges around colonies where per capita food resource is lower. The data also suggest that, over the range of values experienced, the effect of food abundance is greater than the effect of colony size, and that colony size effects may not be evident if food abundance is high.

15. These results predict that foraging range will tend to be small at colonies where food abundance allows high breeding success, but will be greater at colonies where breeding success is reduced by low food abundance. This suggests that long foraging ranges observed in kittiwakes and auks at some colonies in northern Scotland where breeding success is consistently poor due to lack of forage fish (in that example, sandeels), are not applicable to colonies in east England, where such shortages and poor breeding success have not normally been recorded during the JNCC monitoring period of 1986 to 2013 (see summary in Table A1).

Table A.1. Productivity (mean number of chicks per nest) for seabirds at colonies monitored in the JNCC Seabird Monitoring Programme in eastern England and in eastern Scotland (Shetland to Berwickshire) as reported in Annual Reports published by JNCC for the years 1986 to 2006.

Species	Productivity (data from JNCC Annual Reports 1986-2006)		Excess in productivity in E England compared to E Scotland (mean chicks per pair)	Excess in E England productivity as % of E Scotland baseline
	E England Mean (n)	E Scotland Mean (n)		
Gannet	0.77 (8)	0.66 (60)	+0.11	17%
Kittiwake	0.95 (21)	0.67 (84)	+0.28	42%
Common guillemot	0.74 (5)	0.65 (123)	+0.09	14%
Razorbill	0.69 (11)	0.64 (52)	+0.05	8%
Puffin	0.76 (17)	0.62 (49)	+0.14	23%

16. Daunt et al. (2002) point out that seabirds, as central place foragers, will have an upper limit set to their potential foraging range from the colony that is set by time constraints; they assess this to be a limit of 73 km for the kittiwake based on foraging flight speed and time required to catch food as observed for birds from the Isle of May. Kittiwakes would be unable to consistently travel more than 73 km from the colony and provide enough food to keep chicks alive. Hamer et al. (1993) recorded a foraging range exceeding 40 km in 1990 when sandeel stock biomass was very low and breeding success at the study colony in Shetland was 0.0 chicks per nest, but <5 km in 98% of trips in 1991 when sandeel abundance was higher and breeding success was 0.98 chicks per nest. Kotzerka et al. (2010) reported a maximum foraging range of 59 km, with a mean range of around 25 km for a kittiwake colony in Alaska. RSPB's FAME studies have shown some extremely long foraging ranges for seabirds, but those extreme values tend to occur at colonies where food supply is extremely poor and breeding success is low (for example Orkney and Shetland).

17. Data for some seabirds in Shetland show that foraging ranges are now much greater (Wade et al. 2014) than they were in the decades when the sandeel stock at Shetland was large (1970s, early 1980s), and that breeding seabirds make fewer but longer foraging trips and fail to keep chicks alive as a consequence (Heubeck and Parnaby 2012), further indicating the importance of food abundance in determining foraging ranges of breeding seabirds. Chivers et al. (2012) found this same relationship between foraging range from the colony, food abundance and breeding success in kittiwakes at colonies in Northern Ireland. Brown and Grice (2005) report that few common guillemots from the Flamborough colony bring fish back from more than 30 km distant from the colony, consistent with the high breeding success and growing breeding numbers at that colony.

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