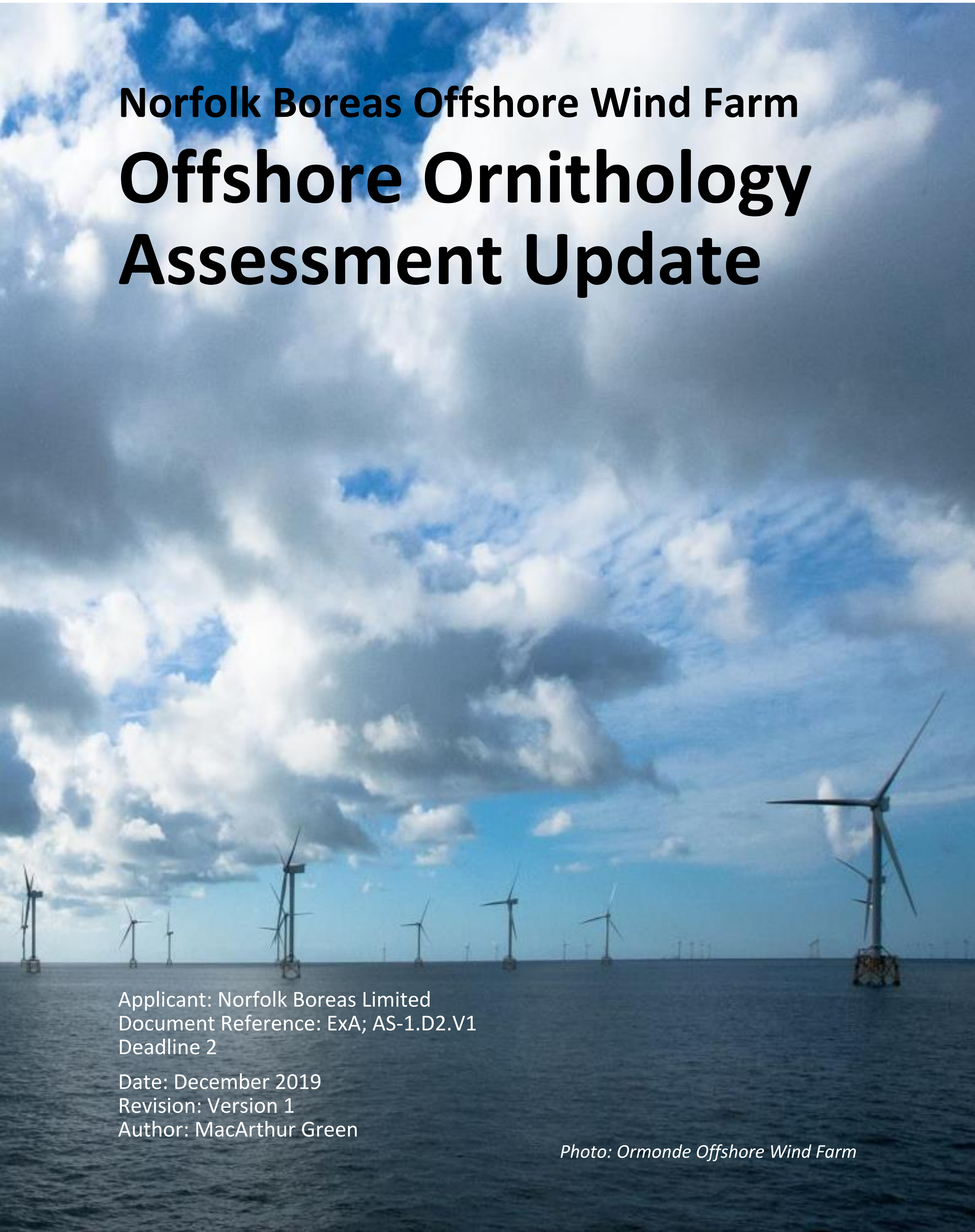
Norfolk Boreas Offshore Wind Farm

Offshore Ornithology Assessment Update



Applicant: Norfolk Boreas Limited
Document Reference: ExA; AS-1.D2.V1
Deadline 2

Date: December 2019
Revision: Version 1
Author: MacArthur Green

Photo: Ormonde Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
07/11/19	01D	First draft for Natural England review	MT	BF/EV	EV
09/12/2019	02D	Update following receipt of Natural England comments	MT	EV	JK
09/12/2019	01F	Final for submission at Deadline 2	MT	EV	JK

Table of Contents

List of Tables

Cited documents

Six documents that were submitted to the Norfolk Vanguard examination are cited within this report and included at the end of this document. These are:

- Red-throated diver displacement. Cited in this document as MacArthur Green 2019d;
- Operational Auk and gannet displacement update and clarification. Cited in this document as MacArthur Green 2019c;
- Offshore Ornithology Cumulative and In-combination Collision Risk Assessment Update. Cited in this document as Norfolk Vanguard 2019;
- Alde Ore Estuary SPA PVA Responses. Cited in this document as MacArthur Green 2019b;
- Precaution in ornithological assessment for offshore wind farms. Cited in this document as MacArthur Green 2019a; and
- Natural England's Comments on Offshore Ornithology Final D8. Cited in this document as Natural England 2019b

Glossary of Acronyms

AEoI	Adverse Effect on Integrity
AOE	Alde-Ore Estuary
BDMPS	Biologically defined minimum population scale
CPS	Counterfactual of Population Size
CRM	Collision Risk Model
CPGR	Counterfactual of Population Growth Rate
EIA	Environmental Impact Assessment
ES	Environmental Statement
FFC	Flamborough and Filey Coast
GB	Great Britain
GIS	Geographical Information System
HRA	Habitats Regulations Assessment
IUCN	International Union for Conservation of Nature and Natural Resources
JNCC	Joint Nature Conservation Committee
LSE	Likely Significant Effect
MW	Mega Watt
NE	Natural England
PEIR	Preliminary Environmental Impact Report
PVA	Population Viability Analysis
RSPB	Royal Society for the Protection of Birds
sCRM	Stochastic Collision Risk Model
SPA	Special Protection Area
UK	United Kingdom

Glossary of Terminology

Export Cables	Cables that transmit power from an offshore electrical platform to the onshore project substation
Norfolk Boreas Site	The Norfolk Boreas wind farm boundary. Located offshore, this will contain all the wind farm array.
Offshore cable corridor	The corridor of seabed from the Norfolk Boreas site to the landfall site within which the offshore export cables will be located.
The Applicant	Norfolk Boreas Limited

1 Executive Summary

This report provides an updated offshore ornithology assessment for the Norfolk Boreas Wind Farm, following receipt of comments and requests for additional assessment from Natural England through their Relevant Representation (RR-099).

The main comments received from Natural England and which have been addressed in this report were with respect to the cumulative and in-combination figures for other wind farms (to review these values, include additional sites and update as necessary), present additional assessment for the project alone taking into account uncertainty in density estimates, consider a wider range of apportioning rates for Special Protection Area (SPA) species, combined collision risk and displacement for gannet and updated population models for gannet, kittiwake and great black-backed gull at the North Sea scale.

The Applicant considers that many of the updates requested by Natural England have added further layers of precaution to what was already a precautionary assessment (APP-201 and APP-226). For example, apportioning of up to 100% of kittiwake on Norfolk Boreas in the breeding season to the Flamborough and Filey Coast SPA, the inclusion of the Hornsea Project Four wind farm (which has only submitted a Preliminary Environmental Impact Report) in the cumulative assessment, consideration of project alone impact magnitudes using the upper 95% confidence density estimates and combining displacement mortality and collision mortality for gannet. These and other aspects of the over-precautionary approaches advised by Natural England were discussed in MacArthur Green (2019a). Among other aspects, this highlighted the extremely low probabilities which should be associated with predictions made following Natural England advice.

The conclusions of the assessment presented in this update remain the same as those in the original application Environmental Statement, ES (APP-226) and Habitats Regulations Assessment, HRA (APP-201).

In summary, the conclusions of the Environmental Impact Assessment (EIA) for the project alone, based on the further layers of precaution mentioned above, are:

- **Collision risks for gannet, kittiwake, lesser black-backed gull, herring gull, great black-backed gull and little gull**, **displacement risks for gannet, razorbill, guillemot and red-throated diver** and **combined collision and displacement for gannet** would not increase the background mortality rate of the relevant populations by more than 1% and hence the magnitudes of effect would be undetectable and **not significant in EIA terms** (sections 4.1.1, 4.2.1, 4.3.1, 4.4.1, 4.4.3.1).

The conclusions of the EIA for the project cumulatively with other plans and projects are:

- **Collision risk for herring gull** and **displacement risk for gannet** would not increase the background mortality rate of the relevant populations by more than 1% and hence the magnitude of cumulative impacts would be undetectable and **not significant in EIA terms** (section 3.4.1, 4.4.1);
- **Collision risks for gannet, kittiwake, lesser black-backed gull and great black-backed gull** and **combined collision and displacement for gannet** could increase the background mortality rate of the relevant populations by more than 1%. However further assessment using population viability analysis (PVA) models found that the predicted worst case magnitudes of cumulative impact would not result in significant impacts on the relevant populations and hence the effects are **not significant in EIA terms** (sections 3.1.1, 3.2.1, 3.3.1, 3.5.1, 4.4.3.3); and
- **Displacement risk for red-throated diver, guillemot and razorbill** could increase the background mortality rate of the relevant populations by more than 1%, however the worst case impact magnitudes are considered to include a large degree of over precaution (i.e. mortality may be over-estimated by up to 10 times) and therefore on the basis of more appropriate (but still precautionary) assessment, the magnitudes of cumulative impact would be undetectable and **not significant in EIA terms** (sections 4.1.2, 4.2.2.1, 4.3.2.1).

The conclusions of the HRA for the project alone are:

- **Collision risks for gannet, kittiwake and little gull** and **displacement risk for razorbill and common scoter** would not increase the background mortality rate of the relevant SPA populations by more than 1% and hence the magnitudes of effect would **not result in Adverse Effects on the Integrity (AEoI)** of the relevant Special Protection Area (SPA) populations (APP-226 and sections 3.2.2, 3.6.1, 4.3.1.1.6, 4.5.1); and
- **Collision risk for lesser black-backed gull**, **displacement risk for gannet and guillemot** and **combined collision and displacement for gannet** could increase the background mortality rate of the relevant populations by more than 1%, however further assessment using PVA models found that the predicted worst case magnitudes of impact on the relevant SPAs population would **not result in AEoI** of the relevant SPAs (section 3.3.2, 4.4.1.1.5, 4.2.1.1.4, 4.4.3.2).
- **Impacts on the seabird assemblage feature** of the **Flamborough and Filey Coast SPA** due to Norfolk Boreas alone **would not result in AEoI** (section 4.6.1).

The conclusions of the HRA for the project in-combination with other plans and projects are:

- **Collision risk for little gull** and **displacement risk for common scoter** would not increase the background mortality rate of the relevant SPA populations by more than 1% and hence the magnitude of effect would **not result in AEoI** of the SPA populations (sections 3.6.2 and 4.5.2); and

- **Collision risks for gannet, kittiwake and lesser black-backed gull** and **displacement risks for red-throated diver, gannet, guillemot and razorbill** and **combined collision and displacement for gannet** could increase the background mortality rate of the relevant populations by more than 1%, however further assessment using PVA models found that the predicted worst case magnitudes of impact on the relevant SPA populations would **not result in AEoI** (sections 3.1.2, 3.2.3, 3.3.3, 4.1.2, 4.2.2.2, 4.3.2.2, APP-201).
- **Impacts on the seabird assemblage feature** of the **Flamborough and Filey Coast SPA** due to Norfolk Boreas in-combination with other plans and projects **would not result in AEoI** (section 4.6.2).

2 Introduction

1. This report has been prepared by MacArthur Green on behalf of Norfolk Boreas Wind Farm Limited. It provides an update of the offshore ornithology assessment for the Norfolk Boreas Wind Farm (presented in APP-201 and APP-226) following receipt of Relevant Representation comments from Natural England (RR-099) and further engagement with Natural England to discuss and address ongoing concerns. As requested by Natural England, this work has been produced as early in the Examination phase as possible to maximise the time available to agree final positions with Natural England. The update contains the following sections:

- Updated collision impact assessment (section 3) for the project alone and cumulatively (Environmental Impact Assessment, EIA) and project alone and in-combination (Habitats Regulations Assessment, HRA) for each species and impact for which this was requested by Natural England;
    - Gannet (EIA cumulative, HRA in-combination),
    - Kittiwake (EIA cumulative, HRA project alone, HRA in-combination),
    - Lesser black-backed gull (EIA cumulative, HRA project alone, HRA in-combination),
    - Herring gull (EIA cumulative),
    - Great black-backed gull (EIA cumulative), and
    - Little gull (HRA project alone, HRA in-combination).
- Updated displacement impact assessment (section 4) for the project alone and cumulatively (EIA) and project alone and in-combination (HRA) for each species and impact for which this was requested by Natural England;
    - Red-throated diver (EIA project alone, EIA cumulative,),
    - Guillemot (EIA project alone, EIA cumulative, HRA project alone, HRA in-combination),
    - Razorbill (EIA project alone, EIA cumulative, HRA project alone, HRA in-combination),
    - Gannet (EIA project alone, EIA cumulative, HRA project alone, HRA in-combination), and
    - Common scoter (HRA project alone and HRA in-combination).
- Update to the assessment of combined collision risk and displacement for:
    - Gannet (EIA project alone, EIA cumulative, HRA project alone, HRA in-combination, section 4.4.3).

2. Furthermore, additional analysis in support of the assessment has been provided in appendices to this report:

- Appendix 1: Analysis of kittiwake age data reported in aerial surveys.

- Appendix 2: Complete cumulative tables of collision risk and displacement for the species listed above.
- Appendix 3: Parameters used in the updated population viability analysis (PVA) undertaken using the recently developed PVA tool developed for Natural England for the following species:
    - Gannet (North Sea scale);
    - Kittiwake (North Sea scale);
    - Lesser black-backed gull (North Sea scale);
    - Great black-backed gull (North Sea scale); and
    - Guillemot (Flamborough and Filey Coast SPA, in-combination scale).

3. Only those sections of the assessment for which Natural England (RR-099) requested an update (e.g. to the methods used) have been provided. For all other aspects of the assessment the original submissions (Norfolk Boreas 2019a APP-201 and Norfolk Boreas 2019b APP-226) remain valid.

2.1 Summary of Natural England Relevant Representation comments

4. Table 2.1 provides a high level list of the Natural England headline comments made in their Relevant Representation (RR-099) and a summary of how these have been addressed. In addition to the updated assessment contained in this report which address the points raised, detailed responses to Natural England's comments are provided in the ExA.RR.D1.V1 (Norfolk Boreas Comments on Relevant Representations).

Table 2.1 Natural England Relevant Representation comments, summary response and guide to section where details are provided.

Natural England headline comment	Response
Breeding season apportionment of impacts for kittiwake and lesser black-backed gull in Habitats Regulations Assessment (HRA)	Breeding season apportioning for kittiwake using a range of values is presented in section 3.2.2. Presentation and discussion of kittiwake age estimates from survey data are provided in Appendix 1 – Analysis of kittiwake age ratios from aerial survey data. The Applicant provided lesser black-backed gull apportioning in the original assessment with rates of up to 30%, which was the upper value requested by Natural England following their review of the draft Habitats Regulations Assessment. Therefore, the Applicant does not consider that additional apportioning ranges are required for this species. The Applicant has discussed this with Natural England, who have confirmed that the range of apportioning rates presented covers the requested range.
Calculation of gannet colony baseline mortality in HRA	The gannet assessment has been updated to provide the additional assessment requested (section 3.1.2)
Consideration of range of predicted impacts due to variability (uncertainty) in EIA and HRA assessments	Consideration of upper and lower 95% confidence intervals for the requested project alone assessments have been provided for kittiwake (section 3.2.2), little gull (section 3.6.1), red-throated

Natural England headline comment	Response
	diver (section 121), guillemot (section 4.2), razorbill (section 4.3) and gannet (section 3.1).
Assessment of displacement impacts	Additional assessment is provided for red-throated diver (sections 121), guillemot (section 4.2) and razorbill (section 4.3). These provide revised and additional estimates for other wind farms and assessment of total values (cumulative and in-combination) with and without the inclusion of Hornsea Project Three and Hornsea Project Four, as requested by Natural England.
Collision risk modelling (CRM) and input parameters	Additional assessment is provided for gannet (section 3.1), kittiwake (section 3.2), lesser black-backed gull (section 3.3), herring gull (section 3.4), great black-backed gull (section 3.5) and the complete cumulative and in-combination tables are provided in Appendix 2 – Cumulative and in-combination assessment tables. It should be noted that this update does not include stochastic collision mortality estimates, as generated using the Marine Scotland stochastic collision risk model (sCRM), because the Applicant has identified further errors in the outputs from this tool (when compared with those obtained from the Band 2012 deterministic method). These errors have been brought to the attention of Marine Scotland Science and the model developer but have not currently been addressed. However, it is important to stress that the assessment follows current best practice (i.e. the Band model) and the conclusions drawn on the outputs remain robust. Furthermore, the mean collision estimates presented in the original assessments (Norfolk Boreas 2019a (APP-201) and Norfolk Boreas 2019b (APP-226)) and in this updated assessment will be the same as those obtained using the sCRM (once the apparent errors in the latter have been addressed). In addition, the upper and lower confidence intervals as presented in the original and updated assessments are also expected to provide a robust guide to the distribution of outputs that will be obtained from the sCRM, although it is acknowledged that inclusion of other sources of uncertainty may increase the confidence interval range. Consequently the conclusions based on the existing collision estimates remain robust and appropriate.
Cumulative and in-combination assessments (displacement and CRM)	Additional assessment is provided for gannet (section 4.4.3), red-throated diver (section 121), kittiwake (section 3.2), lesser black-backed gull (section 3.3), herring gull (section 3.4), great black-backed gull (section 3.5), little gull (section 3.6), razorbill (section 4.3) and guillemot (section 4.2). These provide revised and additional estimates for other wind farms and assessment of total values (cumulative and in-combination) with and without the inclusion of Hornsea Project Three and Hornsea Project Four, as requested by Natural England.
Additive impacts (collision plus displacement for gannet)	Additional assessment is provided for gannet combined collision and displacement in section 4.4.3.
Population modelling (EIA and HRA)	PVA outputs have been updated using the Natural England PVA tool for gannet (sections 3.1 and 4.4.3), for kittiwake (section 3.2.1), for lesser black-backed gull (section 3.3.1), for great black-backed gull (section 3.5.1) and guillemot (section 4.2.2). With the

Natural England headline comment	Response
	exception of guillemot, these all relate to population models at the wider scale as used in the cumulative impact assessment. For guillemot an updated Flamborough and Filey Coast SPA PVA model is presented for the in-combination assessment (this was necessary to extend the range of impacts modelled). Details of the model inputs used with the Natural England PVA are provided in Appendix 3 – NE Seabird PVA Input parameters (gannet), 8.2 (kittiwake), 8.3 (great black-backed gull) and 8.5 (guillemot).

3 Collision Risk Assessment

5. Collision risk assessments have been produced following Natural England advice (RR-099) and the cumulative collision assessments presented in the original application (APP-201 and APP-226) have been updated accordingly. However, the Applicant considers that the methods requested by Natural England, and used for these updated assessments, are over-precautionary and result in greatly over-estimated impacts with highly improbable outcomes. MacArthur Green (2019a) presented detailed discussion on over-precaution in offshore wind farm ornithology assessments, and the key aspects with respect to collision assessments are summarised below:

- **Use of collision estimates calculated for consented wind farm designs in the cumulative and in-combination totals.** Most wind farms have obtained consent for designs based on a large number of small dimension turbines, but then been built with fewer, larger turbines. The consequence of this is that the predicted mortality for the consented project overestimates the predicted mortality for the wind farm that is actually built. Work conducted to investigate this (Trinder 2017, unpublished report) found that the reduction in predicted mortality was around 40% when this straightforward adjustment was made. Examples of this difference are: Dudgeon, assessed number of turbines 168, actual installed number 67; Galloper, assessed 140, installed 56; Race Bank, assessed 116, installed 91; Beatrice, assessed 140, installed 84; Westermost Rough, assessed 50, installed 35. While the installed turbines are usually larger than the assessed ones, partially offsetting the reduction in collision risk, overall the predicted mortalities are always lower following these design updates.
- **Over-estimated nocturnal activity**. Most wind farm collision assessments have used over-estimates of nocturnal activity. For example, for gannet it was previously assumed that this species was 25% as active at night as during the day, and hence the number of flights per 24 hours was found by multiplying the number derived from the daytime survey data by 1.25. Recent work has reported values derived from tagging studies of 3% in the nonbreeding season

and 8% in the breeding season (Furness et al. 2018). Similar magnitudes of reduction have been found in preliminary work for kittiwake, lesser black-backed gull, herring gull and great black-backed gull. Adjustment for these lower nocturnal activity levels reduces collision risks by up to 30%. While Natural England has acknowledged this source of precaution (reflected in requests for project collision estimates with reduced nocturnal rates for gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull) Natural England has not accepted the arguments that this adjustment is equally applicable to consented wind farms included in the cumulative assessments and should therefore be applied retrospectively to update the cumulative and in-combination totals.

- **Over-emphasis on predictions using upper 95% confidence intervals**. For project alone impacts Natural England requested additional assessment using the 95% confidence intervals, alongside that based on the average values. If Natural England considers that the resulting assessment is non-significant for the mean value but significant using the upper 95% estimate, Natural England then concludes that there is low confidence in the conclusion of non-significance. However, this fails to acknowledge the low likelihood (2.5%, equivalent to 1 in 40 probability) of the upper prediction occurring. It is also of note that displacement assessments are conducted on a (biological) seasonal basis using the highest monthly value for the months which fall within a given season. Thus, the 95% confidence intervals are those for the highest abundance, rather than around the mean of the months within that season. This also introduces the potential for double counting across wind farms in cumulative and in-combination assessments since the same individuals may have been recorded (and thereby contribute to the peak estimates) in different wind farms in different months.
- **Slower flight speeds for kittiwake.** Recent studies have reported slower flight speeds for kittiwake (e.g. approx. 9 m/s^{-1}; Skov et al. 2018) compared with the value which has previously been assumed for use in collision modelling (13.1m/s^{-1}). Reducing the value for flight speed entered in the collision model reduces the predicted number of collisions (e.g. by around 15% for a reduction from 13.1ms^{-1} to 9ms^{-1}), and this would apply to all previous wind farm estimates in the same manner as the reduced nocturnal activity.
- **Under estimated avoidance rates.** There is evidence that for some species the currently advised avoidance rates are too low. Bowgen and Cook (2018) used the data collected and reported in Skov et al. (2018) to derive an empirical avoidance rate for kittiwake of 99.0% (which reduces collision estimates by 10% compared with the current value of 98.9%) and 99.5% for gannet (which reduces collision estimates by 55% compared with the current value of 98.9%).

6. If the above sources of precaution in collision modelling are considered together (reduction of c. 40% for built vs. consented wind farms; reduction of c. 30% for lower nocturnal activity rates; reduction of c. 15% for slower kittiwake flight speed and reductions of 55% and 10% respectively for higher gannet and kittiwake avoidance rates), the collision estimates for large gulls will be reduced to around 42% (0.6 x 0.7) of the precautionary values presented in this assessment, with even larger equivalent reductions for kittiwake of around 32% (0.6 x 0.7 x 0.85 x 0.9), and gannet of around 19% (0.6 x 0.7 x 0.45).

7. These clearly represent very significant sources of over precaution in the approach taken to assess offshore wind farm impacts on seabirds and this is particularly apparent in the cumulative and in-combination assessments where the over-precaution in each wind farm assessment is added together. As a consequence of the above concerns, in many instances the conclusions of the updated assessments are considered to greatly over-estimate impact magnitudes and present highly improbable outcomes. The Applicant has therefore highlighted those aspects of the assessment which are considered to be over-precautionary.

3.1 Gannet

3.1.1 EIA Cumulative collisions

8. Updated cumulative collisions for gannet at offshore wind farms in the North Sea and Channel is provided in Table 7.1 (Appendix 2 – Cumulative and in-combination assessment tables). This updated assessment addresses comments provided by Natural England in their Relevant Representation (RR-099), including:

- Estimates for two additional wind farms (Kentish Flats Extension and Methil).
- A review of the figures for all other wind farm sites to ensure consistency with other project assessments (updated values are presented for Dogger Bank Creyke Beck A and B, Hornsea Project Three, Norfolk Vanguard, East Anglia TWO and East Anglia ONE North; the latter two updated from Preliminary Environmental Impact Report to final submission).
- Presentation of total collisions for the following scenarios:
    - With all wind farms (i.e. including all projects at PEIR stage or later);
    - Without the values for the Hornsea Project Three wind farm (as advised by Natural England);
    - Without the values for the Hornsea Project Four wind farm (which has not submitted a final assessment and is therefore currently at PEIR stage); and
    - Without both the Hornsea Project Three wind farm and the Hornsea Project Four wind farm.

9. The total collisions for the scenarios described above are summarised in Table 3.1.

Table 3.1 Summary gannet cumulative collisions.

Cumulative scenario	Breeding season	Autumn migration	Spring migration	Annual
Total (all projects)	1884.2	898.8	373.9	3156.9
Total (minus Hornsea Project Three)	1858.2	886.9	362.9	3107.9
Total (minus Hornsea Project Four)	1841.2	888.7	365.8	3095.6
Total (minus Hornsea Projects Three and Four)	1815.2	876.7	354.9	3046.6

10. The cumulative annual total collisions including all projects is 3,157, which reduces to 3,108 with Hornsea Project Three omitted and reduces further to 3,047 with the additional omission of Hornsea Project Four.

11. The largest gannet BDMPS is 456,298 and the biogeographic population is 1,180,000. The background mortality for these populations, calculated using an all age class mortality rate of 0.191 (see Table 13.13 in APP 226 for details of how this was calculated), are 87,153 and 225,380 respectively. The addition of 3,047 to 3,157 to these increases the background mortality by between 3.5% and 3.6% (BDMPS) and between 1.3% and 1.4% (biogeographic), taking into account the incorporation of precautionary aspects as noted above (section 3).

12. Natural England requested that the PVA outputs for this species were updated using the recently developed NE PVA Tool (https://github.com/naturalengland/Seabird_PVA_Tool). The model settings are provided in Section 8.1 and the results from the PVA are presented in Table 3.2.

Table 3.2 Gannet BDMPS and biogeographic population modelling results using the NE PVA tool

Population scale	Adult mortality	Density independent counterfactual metric (after 30 years)		Density dependent counterfactual metric (after 30 years)	
		Growth rate	Population size	Growth rate	Population size
BDMPS	3000	0.9923	0.7867	0.9924	0.7885
	3100	0.9920	0.7805	0.9921	0.7824
	3200	0.9918	0.7744	0.9919	0.7761
Biogeographic	3000	0.9970	0.9116	0.9971	0.9125
	3100	0.9969	0.9087	0.9970	0.9097
	3200	0.9968	0.9059	0.9969	0.9070

13. Although both counterfactual measures (of population size and population growth rate) are presented in Table 3.2, the Applicant considers that the counterfactuals of

population growth rate are more informative and credible for assessment purposes for the following reasons:

- The counterfactual of population growth rate can be compared to recent and longer-term population trends and represents a measure of the population's resilience and ability to regenerate. It is also relatively insensitive to the absolute value for the baseline rate of growth or direction (positive or negative);
- In contrast the counterfactual of population size is much more sensitive to the predicted population trend (strength of growth and direction). This is particularly true in the absence of density dependence. For example, a population with a positive growth rate will grow exponentially, with the result that very large differences can be obtained in the baseline (unimpacted) population size and impacted population size (neither of which prediction is credible since seabird populations are constrained by factors such as nest site availability, prey availability, etc.).

14. For these reasons the interpretation of PVA outputs focusses on the counterfactuals of population growth rate.

15. The maximum reduction in the BDMPS population growth rate at an adult mortality of 3,200, using the more precautionary density independent model, was 0.82% (0.9918). The equivalent reduction for the biogeographic population was 0.32% (0.9968). Similar reductions were obtained using the density dependent model.

16. These compare to the observed rate of 2.9% at which the Scottish population (which holds the majority of the British population) has grown over the last 15 years (Murray et al. 2015). While the UK gannet population is reported to have increased by 34% between 2003/4 and 2015[1] (an annual rate of growth of approximately 1.4%). Gannet are classed as 'least concern' by the International Union for Conservation of Nature and Natural Resources (IUCN), the lowest level of concern in the IUCN Red List grading system.

17. Thus, assessed against both the BDMPS and the biogeographic population the number of predicted collision mortalities at North Sea and Channel wind farms is not at a level which would trigger a risk of population decline, but would only result in a slight reduction in the current growth rates, despite the incorporation of precautionary aspects as noted above (section 3). Consequently, the cumulative impact on the gannet population due to cumulative collisions is considered to be of low magnitude and the impact significance is minor adverse.

[1] https://jncc.gov.uk/our-work/northern-gannet-morus-bassanus/#annual-abundance-and-productivity-by-geographical-area-united-kingdom

3.1.2 HRA In-combination

18. The in-combination gannet collisions apportioned to the FFC SPA are presented in full in Table 7.1 (Appendix 2 – Cumulative and in-combination assessment tables), and summarised for the different scenarios (see section 3.1.1) in Table 3.3.

Table 3.3 Summary gannet in-combination collisions apportioned to the FFC SPA.

cumulative scenario	Breeding season	Autumn migration	Spring migration	Annual
Total (all projects)	336.5	43.1	23.2	402.8
Total (minus Hornsea Project Three)	310.5	42.6	22.5	375.6
Total (minus Hornsea Project Four)	293.2	42.7	22.7	358.6
Total (minus Hornsea Projects Three and Four)	267.2	42.1	22.0	331.3

19. The in-combination annual total collisions including all projects is 403, which reduces to 376 with Hornsea Project Three omitted and is further reduced to 331 with the additional omission of Hornsea Project Four.

20. The FFC SPA population at designation was 22,122 breeding adults, although it was more recently counted at 26,782 (Aitken et al. 2017). The background mortality for these population sizes, calculated using the adult mortality rate of 0.08, are 1,770 and 2,142 respectively. Addition of highly precautionary estimates of between 331 and 403 to these increases the background mortality by between 18.7% and 22.7% (designated) or by between 15.4% and 18.8% (2017 count).

21. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for mortality levels of 325 and 400 (the nearest values to the project alone and in-combination predictions) are provided in Table 3.4.

Table 3.4 Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	325	0.985	0.651	Tables A2_1.1 & A2_1.3
	2		0.985	0.651	Tables A2_3.1 & A2_3.3

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density dependent	1		0.991	0.739	Tables A2_2.1 & A2_2.3
	2		0.990	0.738	Tables A2_4.1 & A2_4.3
Density independent	1	400	0.982	0.589	Tables A2_1.1 & A2_1.3
	2		0.982	0.589	Tables A2_3.1 & A2_3.3
Density dependent	1		0.988	0.685	Tables A2_2.1 & A2_2.3
	2		0.988	0.683	Tables A2_4.1 & A2_4.3

22. As noted above (paragraph 13), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

23. The maximum reduction in the population growth rate, at a highly precautionary adult mortality of 400 using the more precautionary density independent model was 1.8% (0.982).

24. This compares to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. A reduction of less than 2% in this case represents a negligible risk for the population. Natural England (2019b) suggested that, if the SPA population follows a similar pattern of growth to those observed at colonies of a similar age, the observed rate of growth is likely to decrease over the coming decades. Natural England (2019b) does not discuss the reasons for this apparent pattern in other colonies, however it is reasonable to assume that this would occur due to increasing levels of competition for resources, in other words density dependence. On this basis the results from the density dependent PVA are appropriate. These indicate a maximum growth rate reduction of 1.2% (0.988 at a mortality of 400), which is still expected to be much less than the overall rate of population growth during the lifetime of the Norfolk Boreas wind farm and thus still represents a negligible risk to the population.

25. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

26. On the basis of the population model predictions the number of predicted collision mortalities at Norfolk Boreas in-combination with other North Sea wind farms attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger

a risk of population decline, but would only result in a slight reduction in the growth rate currently seen at this colony.

27. Therefore, it can be concluded that, even with the high degree of precaution in the assessment, there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on gannet due to in-combination collision mortality.

3.2 Kittiwake

3.2.1 EIA Cumulative collisions

28. Updated cumulative collisions for kittiwake at offshore wind farms in the North Sea and Channel is provided in Table 7.2 (Appendix 2 – Cumulative and in-combination assessment tables). The update addresses comments provided by Natural England in their Relevant Representation (RR-099), including:

- Estimates for two additional wind farms (Kentish Flats Extension and Methil).
- A review of the figures for all other wind farm sites to ensure consistency with other project assessments (updated values are presented for Dogger Bank Creyke Beck A and B, Hornsea Project Three, Norfolk Vanguard, East Anglia TWO and East Anglia ONE North; the latter two updated from Preliminary Environmental Impact Report to final submission).
- Presentation of totals:
  - With all wind farms (i.e. including all projects at PEIR stage or later);
  - Without the values for the Hornsea Project Three wind farm (as advised by Natural England);
  - Without the values for the Hornsea Project Four wind farm (which has not submitted a final assessment and is therefore currently at PEIR stage); and
  - Without both the Hornsea Project Three wind farm and the Hornsea Project Four wind farm.

29. The total collisions for the four scenarios described above are summarised in Table 3.5.

Table 3.5 Summary kittiwake cumulative collisions.

Wind farm cumulative scenario	Breeding season	Autumn migration	Spring migration	Annual
Total (all projects)	1469.9	1704.3	1223.4	4397.7
Total (minus Hornsea Project Three)	1282.4	1609.8	1208.4	4100.6
Total (minus Hornsea Project Four)	1316.6	1669.7	1213.5	4199.8
Total (minus Hornsea Projects Three and Four)	1129.1	1575.1	1198.5	3902.7

30. Taking into account all the levels of precaution listed in Section 3, the cumulative annual total collisions including all projects is 4,397, which reduces to 4,100 with Hornsea Project Three omitted and is further reduced to 3,903 with the additional omission of Hornsea Project Four.

31. The largest kittiwake BDMPS is 829,937 and the biogeographic population is 5,100,000. The background mortality for these populations, calculated using an all age class mortality rate of 0.156 (see Table 13.13 in APP 226 for details of how this was calculated) are 129,470 and 795,600 respectively. Addition of 3,903 to 4,397 to these increases the background mortality by 3.0% to 3.4% (BDMPS) and 0.49% to 0.55% (biogeographic).

32. Natural England requested that the PVA outputs for this species were updated using the recently developed NE PVA Tool (https://github.com/naturalengland/Seabird_PVA_Tool). The model settings are provided in Appendix 3 – NE Seabird PVA Input parameters and the results from the PVA are presented in Table 3.6.

Table 3.6 Kittiwake BDMPS and biogeographic population modelling results using the NE PVA tool.

Population scale	Adult mortality	Density independent counterfactual metric (after 30 years)	
		Growth rate	Population size
BDMPS	3900	0.9944	0.8410
	4000	0.9943	0.8376
	4100	0.9941	0.8335
	4200	0.9940	0.8302
	4300	0.9939	0.8268
	4400	0.9937	0.8229
Biogeographic	3900	0.9991	0.9723
	4000	0.9991	0.9717
	4100	0.9990	0.9711
	4200	0.9990	0.9703
	4300	0.9990	0.9697
	4400	0.9989	0.9688

33. The maximum reduction in the BDMPS population growth rate at an adult mortality of 4,400 using a precautionary density independent model was 0.63% (0.9937). The equivalent reduction for the biogeographic population was 0.11% (0.9989).

34. Although both counterfactual measures (of population size and population growth rate) are presented in Table 3.6, the Applicant considers that the counterfactuals of

population growth rate are more informative and credible for assessment purposes for the following reasons:

- The counterfactual of population growth rate can be compared to recent and longer-term population trends and represents a measure of the population's resilience and ability to regenerate. It is also relatively insensitive to the absolute value for the baseline rate of growth or direction (positive or negative);
- In contrast the counterfactual of population size is much more sensitive to the predicted population trend (strength of growth and direction). This is particularly true in the absence of density dependence. For example, a population with a positive growth rate will grow exponentially, with the result that very large differences can be obtained in the baseline (unimpacted) population size and impacted population size (neither of which prediction is credible since seabird populations are constrained by factors such as nest site availability, prey availability, etc.).

35. For these reasons the interpretation of PVA outputs focusses on the counterfactuals of population growth rate.

36. To place the predicted magnitudes of change in context, over three approximate 15 year periods (between censuses) the British kittiwake population changed by +24% (1969 to 1985), -25% (1985 to 1998) and -44% (2000 to 2015) (http://jncc.defra.gov.uk/page-3201 accessed 18th October 2019). Changes of between 0.11% and 0.63% across a longer (30 year) period against a background of natural changes an order of magnitude larger would almost certainly be undetectable. Kittiwake is classed as 'vulnerable' by the IUCN due to declines over the last three generations which are considered likely to continue.

37. Natural England advised that the results from density independent models should be used where '*there is not sufficient evidence that compensatory density dependence is operating on the population*' (Natural England 2018). Consequently, the outputs presented here were obtained from density independent simulations. However, there is evidence for density dependent regulation of the North Sea kittiwake population, and this evidence was summarised in EATL (2016). While Natural England accepted the conclusions of this review (that there was strong evidence for the presence of density dependence operating in the population) they maintained that because its mode of operation was less clear, the results of the density independent PVA models should be used in preference to the density dependent ones (acknowledging that the results were the worst case). Thus, although estimating robust parameters for the model is extremely difficult, there is good evidence that the North Sea population itself is regulated by density dependent processes.

38. The consequence of this is that the results as presented in Table 3.6 provide precautionary predictions (in addition to those noted in section 3) of the potential effects of cumulative collisions, and yet even on this basis the reduction in the long term growth rate is no more than 0.6%.

39. Kittiwake is considered to be of low to medium sensitivity, low to medium conservation value, the magnitude of effect described above is considered to be low and the relative contribution of Norfolk Boreas to this cumulative total is small. As noted above (section 3), there are several additive sources of precaution in the collision assessment (precautionary avoidance rate estimates, reduction in wind farm sizes and number of turbines, over-estimated nocturnal activity) which mean that the total mortality is almost certainly considerably lower (i.e. by up to 65%) than that based on the precautionary approaches requested by Natural England and presented here, and these have also been assessed using the more precautionary density independent PVA. Consequently, the worst case cumulative collision mortality is considered to be of low magnitude, resulting in impacts of minor adverse significance. However, when the various sources of precaution are taken into account the cumulative collision risk impact magnitude is almost certainly even smaller.

3.2.2 HRA Project alone

40. In the original assessment (APP-201), a review of available evidence was used to make a precautionary estimate that 26.1% of the birds observed on the wind farm during the breeding season were potentially from the SPA. This was based on consideration of a range of data sources, including RSPB tracking data (Wischnewski et al. 2018) and reviews of seabird distributions, age structure and colony exchange rates (e.g. Coulson 2011, Wakefield et al. 2017). In their Relevant Representation (RR-099), Natural England advised that a range of apportioning rates should be used for estimating the potential degree of connectivity between Norfolk Boreas and the FFC SPA, including up to 100%. Table 3.7 provides estimates of the collisions apportioned to the SPA from Norfolk Boreas on this basis, with apportioning rates of 10% to 100% and including the Applicant’s estimate of 26.1%. These are presented for the migration free and full breeding seasons and with the 95% confidence intervals.

Table 3.7 Estimates of kittiwake collisions apportioned to the FFC SPA using Natural England advised range of rates.

Breeding season apportioning percentage	Collisions (migration free breeding season)				Collisions (full breeding season)			
	Breeding season	Autumn	Spring	Annual	Breeding season	Autumn	Spring	Annual
10	3 (0.8-6)	6.3 (3.2-10.3)	4.1 (1.3-7.5)	13.3 (5.3-10.6)	4.2 (1.2-8.4)	6.1 (3.2-9.9)	3.4 (1-6.4)	13.7 (5.5-24.6)
20	6 (1.6-12)			16.3 (6.1-16.6)	8.3 (2.4-16.7)			17.9 (6.7-33)
26.1	*7.8 (2-15.7)*			*18.2 (6.6-20.2)*	*10.9 (3.2-21.8)*			*20.4 (7.4-38.1)*
30	9 (2.3-18)			19.3 (6.9-22.6)	12.5 (3.7-25.1)			22.1 (7.9-41.3)
40	12 (3.1-24)			22.3 (7.7-28.6)	16.7 (4.9-33.4)			26.2 (9.1-49.7)
50	15 (3.9-30)			25.3 (8.4-34.6)	20.8 (6.1-41.8)			30.4 (10.3-58)
60	18 (4.7-36)			28.3 (9.2-40.6)	25 (7.3-50.2)			34.6 (11.6-66.4)
70	20.9 (5.4-42)			31.3 (10-46.6)	29.2 (8.5-58.5)			38.7 (12.8-74.8)
80	23.9 (6.2-48)			34.3 (10.8-52.6)	33.3 (9.8-66.9)			42.9 (14-83.1)
90	26.9 (7-54)			37.3 (11.6-58.6)	37.5 (11-75.2)			47.1 (15.2-91.5)
100	29.9 (7.8-60)			40.3 (12.3-64.6)	41.7 (12.2-83.6)			51.2 (16.4-99.8)

41. Natural England also requested presentation of the kittiwake age estimates made from the aerial survey data. These data, and consideration of their reliability for the current purposes are provided in Appendix 1 – Analysis of kittiwake age ratios from aerial survey data. The review of these data, conducted in relation to the known plumage characteristics of this species, found that the survey data do not provide a reliable basis for estimating the proportion of adults and sub-adults present on the Norfolk Boreas site. Crucially, while only a small percentage of kittiwakes were positively identified as sub-adults (juveniles and immatures) it is considered that a large (but unknown) percentage of birds identified as adults are in fact very likely to have been immature birds, and this prevents any further meaningful analysis. Thus, these data are not given any further consideration in relation to the assessment.

42. The annual total collisions (including highly precautionary assumptions, section 3) using a range of breeding season apportioning rates from 10% to 100% is between 13.3 (confidence intervals 5.3-10.6) and 40.3 (confidence intervals: 12.3-64.6) collisions per year for the migration free breeding season and 13.7 (confidence intervals: 5.5-24.6) to 51.2 (confidence intervals: 16.4 - 99.8) for the full breeding season. Using the Applicant’s estimate of the most appropriate apportioning rate (26.1% in the breeding season) the annual mortality prediction using the migration

free breeding season is for 18.2 (confidence intervals: 6.6 – 20.2) and using the full breeding season is 20.4 (confidence intervals: 7.4 – 38.1).

43. The FFC SPA population at designation was 44,520 pairs. The background mortality for this population, calculated using the adult mortality rate of 0.146 (Horswill and Robinson 2015) is 13,000. The 1% threshold of detectability for this population is therefore an additional mortality of 130 individuals. Even the most highly precautionary value of 99.8 (assuming 100% breeding season apportioning, the full breeding season and upper 95% confidence interval) is below this threshold.

44. Thus, even on the basis of a further precautionary assumption, that all kittiwakes observed on the wind farm in the breeding season are breeding birds from the FFC SPA (which is considered very unlikely), the increase in mortality obtained using precautionary collision modelling assumptions would not materially alter the background mortality of the population and would be undetectable. Therefore, there is no risk of an adverse effect on the integrity of the FFC SPA due to collisions at Norfolk Boreas.

3.2.3 HRA In-combination

45. The in-combination kittiwake collisions apportioned to the FFC SPA are presented in full in Table 7.1 (Appendix 2), and summarised below in Table 3.8 for the four scenarios described in Section 3.2.1. Note that the totals in Table 3.8 include Norfolk Vanguard and Norfolk Boreas using the full migration season and a breeding season apportioning rate of 26.1% (as discussed above and in APP-201). In the assessment below consideration has also been given to the totals obtained when Natural England's preferred apportioning rate of 86% is applied to these projects.

Table 3.8 Summary kittiwake in-combination collisions apportioned to the FFC SPA.

Wind farm cumulative scenario	Breeding season	Autumn migration	Spring migration	Annual
Total (all projects)	489.0	92.0	88.1	668.8
Total (minus Hornsea Project Three)	312.7	86.9	86.9	486.4
Total (minus Hornsea Project Four)	335.7	90.1	87.3	512.9
Total (minus Hornsea Projects Three and Four)	159.4	85.0	86.2	330.5

46. The in-combination annual total collisions including all projects is 669, which reduces to 486 with Hornsea Project Three omitted and is further reduced to 330 with the omission of Hornsea Project Four (at PEIR stage).

47. The FFC SPA population at designation was 44,520 pairs. The background mortality for this population, calculated using the adult mortality rate of 0.146 (Horswill and

Robinson 2015) is 13,000. Addition of highly precautionary estimates of between 330 and 669 to this increases the background mortality by 2.5% to 5.1%.

48. If the breeding season apportioning rate for Norfolk Vanguard and Norfolk Boreas is increased to 86% (Natural England's preferred rate) the in-combination totals for all wind farms increases to 728 and the total without Hornsea Projects Three and Four increases to 389.2. Consideration of these magnitudes of potential impact is provided below.

49. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for mortality levels of 300, 400, 650 and 750 are provided in Table 3.9. It should be noted that the maximum level of mortality (750) also includes allowance for the highly precautionary assumption that up to 86% of the breeding season collisions at Norfolk Vanguard and Norfolk Boreas are breeding birds from the SPA.

Table 3.9 Kittiwake FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	300	0.997	0.906	Tables A2_5.1 & A2_5.3
	2		0.997	0.907	Tables A2_7.1 & A2_7.3
Density dependent	1		0.999	0.973	Tables A2_6.1 & A2_6.3
	2		0.999	0.971	Tables A2_8.1 & A2_8.3
Density independent	1	400	0.996	0.877	Tables A2_5.1 & A2_5.3
	2		0.995	0.877	Tables A2_7.1 & A2_7.3
Density dependent	1		0.999	0.963	Tables A2_6.1 & A2_6.3
	2		0.999	0.959	Tables A2_8.1 & A2_8.3
Density independent	1	650	0.993	0.809	Tables A2_5.1 & A2_5.3
	2		0.993	0.809	Tables A2_7.1 & A2_7.3
Density dependent	1		0.999	0.940	Tables A2_6.1 & A2_6.3
	2		0.999	0.936	Tables A2_8.1 & A2_8.3
Density independent	1	750	0.992	0.783	Tables A2_5.1 & A2_5.3
	2		0.992	0.782	Tables A2_7.1 & A2_7.3
	1		0.998	0.931	Tables A2_6.1 & A2_6.3

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density dependent	2		0.998	0.926	Tables A2_8.1 & A2_8.3

50. As noted above (paragraph 34), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

51. The maximum reduction in the population growth rate, at an adult mortality of 750 using the more precautionary density independent model was 0.8% (0.992). Using the more realistic density dependent model the reduction would be 0.2% (0.998).

52. The maximum reduction in growth rate (0.8%; based on highly precautionary assumptions, see section 3), suggests that even this worst case and unrealistic scenario represents only a very small risk to the SPA population.

53. With lower total mortality (e.g. 650), as would be expected using the Applicant's evidence based apportioning rate of 26.1% for Norfolk Vanguard and Norfolk Boreas, the predicted impact is reduced, with a growth rate reduction of no more than 0.7%.

54. The conservation objective for the kittiwake feature of the SPA is to restore the population to the 1987 estimate of 83,700 pairs. The kittiwake breeding numbers at the Flamborough and Filey Coast SPA have remained relatively stable around an average of approximately 40,000 pairs over the last 20 years, so this target would not appear currently to be being achieved (i.e. before any consideration is given to the additional mortality currently predicted, almost all of which is attributable to wind farms which have not yet been constructed). Furthermore, the Applicant notes that there are several compelling reasons to consider that the apparent population estimate in 1987 was recorded in error and in fact represented the estimate of breeding *individuals* and not of *pairs* (e.g. Coulson 2011, 2017). Nonetheless, the RSPB reported that since 2000 the population has grown by 7% (Aitken et al. 2017).

55. Therefore, to conclude this section, although the kittiwake population has grown over the last two decades, suggesting that it could be on track to achieving the stated conservation objective, there is robust scientific evidence that the target objective for this population is in fact erroneous. This would suggest that the population is maintaining itself around a size of 40,000-50,000 pairs.

56. On the basis of the more realistic density dependent population model predictions, and the best guide to future trends available (i.e. the recent trend in the population)

even the highly precautionary upper estimate of the number of in-combination kittiwake collisions attributed to the Flamborough & Filey Coast SPA (750) is not at a level which would trigger a risk of population decline, but would just result in a slight reduction in the growth rate currently seen at this colony.

57. In addition to the assumption that 100% of breeding season collisions on Norfolk Boreas are birds from the FFC SPA, the collision total also includes several other sources of precaution, including over-estimated nocturnal activity for existing projects and the use of consented collision estimates for projects which have since been constructed to designs which will have much lower collision risks (due to reduced rotor swept areas; see section 3).

58. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on kittiwake due to the proposed Norfolk Boreas project in-combination with other plans and projects.

3.3 Lesser black-backed gull

3.3.1 EIA Cumulative collisions

59. Updated cumulative collisions for lesser black-backed gull at offshore wind farms in the North Sea and Channel is provided in Table 7.4 (Appendix 2). The update addresses comments provided by Natural England in their Relevant Representation (RR-099), including:

- Estimates for two additional wind farms (Kentish Flats Extension and Methil).
- A review of the figures for other wind farm sites to ensure consistency with other project assessments (updated values are presented for Hornsea Project Three, Thanet Extension, Norfolk Vanguard, East Anglia TWO and East Anglia ONE North; the latter two updated from Preliminary Environmental Impact Report to final submission).
- Presentation of collision totals for the following scenarios:
    - With all wind farms (i.e. including all projects at PEIR stage or later);
    - Without the values for the Hornsea Project Three wind farm (as advised by Natural England);
    - Without the values for the Hornsea Project Four wind farm (which has not submitted a final assessment and is therefore currently at PEIR stage); and
    - Without both the Hornsea Project Three wind farm and the Hornsea Project Four wind farm.

60. The total collisions for the scenarios described above are summarised in Table 3.10.

Table 3.10 Summary lesser black-backed gull cumulative collisions.

Wind farm cumulative scenario	Breeding season	Nonbreeding season	Annual
Total (all projects)	191.2	390.9	582.1
Total (minus Hornsea Project Three)	173.9	390.9	564.8
Total (minus Hornsea Project Four)	189.3	390.9	580.2
Total (minus Hornsea Projects Three and Four)	172.0	390.9	562.9

61. Taking into account the precautionary sources described in Section 3 the cumulative annual total collisions including all projects is 582, which reduces to 565 with Hornsea Project Three omitted and reduces further to 563 with the additional omission of Hornsea Project Four.

62. The largest lesser black-backed gull BDMPS is 209,007 and the biogeographic population is 864,000. The background mortality for these populations, calculated using an all age class mortality rate of 0.124 (see Table 13.13 in APP 226 for details of how this was calculated) are 25,917 and 107,136 respectively. Addition of 563 to 582 to these increases the background mortality by 2.2% (BDMPS) and 0.5% (biogeographic). Thus, in relation to the biogeographic population this level of mortality would be undetectable. Further consideration is provided in relation to the BDMPS population below.

63. However, for several reasons (listed in Section 3) the cumulative total is considered to be over precautionary with the consequence that the impact magnitude is inflated. It is clear therefore that the current cumulative collision total of 583 is highly precautionary and that with the adjustments noted above (section 3), the estimated increase in background mortality would be much lower. However, at Natural England's request further consideration is provided using an update to the previous population modelling.

64. Natural England requested that the PVA outputs for this species were updated using the recently developed NE PVA Tool (https://github.com/naturalengland/Seabird_PVA_Tool). The model settings are provided in Section 8.3 and the results from the PVA are presented in Table 3.11.

Table 3.11 Lesser black-backed gull BDMPS population modelling results using the NE PVA tool.

Population scale	Adult mortality	Density independent counterfactual metric (after 30 years)	
		Growth rate	Population size
BDMPS	500	0.9973	0.9191
	600	0.9967	0.9035

65. The maximum reduction in the BDMPS population growth rate at an adult mortality of 600 using a precautionary density independent model was 0.33% (0.9967).

66. The UK lesser black-backed gull population increased by 81% between 1969 and 2002 (the only UK wide estimates considered reliable; JNCC.gov.uk). This represents an average annual growth rate of 1.8%, therefore a maximum reduction of 0.33% is not considered likely to result in any significant, nor detectable effects on the BDMPS population and the cumulative impact on the lesser black-backed gull population, assessed against the BDMPS population, resulting from the highly precautionary collision estimates is not at a level which would trigger a risk of population decline but would only result in a slight reduction in the current growth rates. Consequently, the impact on the lesser black-backed gull population due to cumulative collisions is considered to be of low magnitude, for a species which has low to medium sensitivity (classed as 'least concern' by the IUCN) and therefore the impact significance is minor adverse.

67. Furthermore, for several reasons the cumulative total is considered to be over precautionary with the consequence that the impact magnitude is inflated (these include built vs. consented wind farm designs and over-estimates of nocturnal flight activity; see section 3). Incorporation of only these two factors in the collision models reduces predicted collisions by almost 60%.

68. It is clear therefore that the current cumulative collision total of 582 is highly precautionary and that with the application of adjustments, as noted above, the estimated increase in background mortality for the smaller BDMPS population would be reduced to below the 1% threshold for detection (e.g. 582 x consented reduction 0.6 x nocturnal reduction 0.75 = 261 which would equate to a 1% increase in background mortality). Consequently, the cumulative impact on the lesser black-backed gull population due to cumulative collisions is considered to be of low magnitude, so for this low sensitivity species the impact significance is minor adverse.

3.3.2 HRA Project alone

69. Natural England requested that additional assessment be presented giving consideration to uncertainty in the density estimates through the presentation of collision impacts using the upper and lower confidence estimates (obtained using the 95% confidence intervals on density) as well as the mean values and these are presented in Table 3.12.

Table 3.12 Estimates of lesser black-backed gull collisions apportioned to the Alde Ore Estuary SPA using Natural England advised range of rates and including 95% confidence estimates.

Month*	Deterministic collision mortality (mean density and 95% c.i.)	Alde-Ore Estuary SPA collisions (assumed 21% breeding season, 3.3% migration periods and 5% in mid-winter)	Alde-Ore Estuary SPA collisions (assumed 30% breeding season, 3.3% migration periods and 5% in mid-winter)
January	1.67 (0-4.94)	0.08 (0-0.25)	0.08 (0-0.25)
February	0.38 (0-2.32)	0.02 (0-0.12)	0.02 (0-0.12)
March	0.46 (0-2.71)	0.02 (0-0.09)	0.02 (0-0.09)
April	1.45 (0-6.51)	0.3 (0-1.37)	0.44 (0-1.95)
May	1.01 (0-3.03)	0.21 (0-0.64)	0.3 (0-0.91)
June	1.47 (0-6)	0.31 (0-1.26)	0.44 (0-1.8)
July	5.54 (1.02-13.3)	1.16 (0.21-2.79)	1.66 (0.31-3.99)
August	7.83 (2.94-13.76)	1.64 (0.62-2.89)	2.35 (0.88-4.13)
September	16.57 (0-42.37)	0.55 (0-1.4)	0.55 (0-1.4)
October	1.31 (0-5.27)	0.04 (0-0.17)	0.04 (0-0.17)
November	0.82 (0-4.02)	0.04 (0-0.2)	0.04 (0-0.2)
December	1.27 (0-4.06)	0.06 (0-0.2)	0.06 (0-0.2)
Total	39.8 (4.0-108.3)	4.44 (0.83-11.37)	6 (1.19-15.21)

* Note that this assessment applies the full breeding season (April to August).

70. Natural mortality for the SPA population (assuming approximately 4,000 adults) would be around 460 individuals at an average adult mortality rate of 11.5% (Horswill and Robinson 2015). A total additional worst case mean mortality of up to four (using the evidence based breeding season rate of 21%; APP-201) or six (at Natural England's precautionary rate of 30%) birds due to collisions at the Norfolk Boreas site would increase the mortality rate by 0.8% to 1.3%.

71. Considering the evidence based apportioning rate (21%), the 95% confidence intervals are 0.8-11.4 collisions while using Natural England's preferred rate (30%), the 95% confidence intervals are 1.2-15.2 collisions. Thus, the evidence based assessment predicts increases in mortality of between 0.1% and 2.5% and Natural England's preferred approach predicts increases in mortality of between 0.3% and 3.3%.

72. A population model was developed to provide further interpretation of the potential impacts (MacArthur Green 2019b). This model was developed following current NE guidance, utilising a matched-run approach to generate counterfactuals of population size (CPS) and counterfactuals of population growth rate (CPGR) and run for a simulated period of 30 years. Summary results are provided in Table 3.13.

Table 3.13. Lesser black-backed gull Alde Ore Estuary SPA population modelling results (see MacArthur Green 2019 for details).

Model	Adult mortality	Counterfactual metric (after 30 years)	
		Growth rate	Population size
Density independent	10	0.997	0.930
	15	0.996	0.897
Density dependent	10	0.999	0.979
	15	0.999	0.968

73. Although both counterfactual measures (of population size and population growth rate) are presented in Table 3.13, the Applicant considers that the counterfactuals of population growth rate are more informative and credible for assessment purposes for the following reasons:

- The counterfactual of population growth rate can be compared to recent and longer-term population trends and represents a measure of the population's resilience and ability to regenerate. It is also relatively insensitive to the absolute value for the baseline rate of growth or direction (positive or negative);
- In contrast the counterfactual of population size is much more sensitive to the predicted population trend (strength of growth and direction). This is particularly true in the absence of density dependence. For example, a population with a positive growth rate will grow exponentially, with the result that very large differences can be obtained in the baseline (unimpacted) population size and impacted population size (neither of which prediction is credible since seabird populations are constrained by factors such as nest site availability, prey availability, etc.).

74. For these reasons the interpretation of PVA outputs focusses on the counterfactuals of population growth rate.

75. Taking the modelled adult mortality of 15 (as the worst case), the population growth rate was predicted to be 0.4% lower (0.996) than the baseline using the density independent model, and 0.1% lower (0.999) using the density dependent model. At the lower modelled adult mortality of 10, the reduction in growth rate was 0.3% for the density independent model and 0.1% for the density dependent model.

76. Although there is a lack of reliable evidence on the population trend at the SPA since 2010 (the last all SPA count available), the predicted reductions in growth rate, which are all much less than 1% even at a mortality of 15, are considered very unlikely to have a detectable effect on the population.

77. The conservation objective is to restore the population to a size greater than 14,074, which was the estimated size between 1994 and 1997. The population in fact

continued to grow until 2000 (reaching a peak of over 20,000 pairs) but then crashed to almost 5,000 in 2001. It is thought that this was triggered by several farming changes in the region, with an outbreak of swine fever in 2000 and the further loss of pig farms due to the foot and mouth epidemic in 2001. Following this the population continued to decline with the most recent SPA population estimate of 1,940 pairs (2011-2015). Clearly, none of this decline was related to offshore wind farm impacts since none were constructed at these times. At the same time there appears to have been large increases in the urban gull populations, which may potentially be connected to the changes at the SPA. Against this background, it is clear that the potential mortality of lesser black-backed gull due to Norfolk Boreas, even when estimated with the high levels of precaution presented here (see section 3 and based on the density independent PVA outputs), is very small and will not result in an adverse effect on the integrity of the Alde-Ore Estuary SPA.

3.3.3 HRA In-combination

78. The in-combination lesser black-backed gull collisions apportioned to the Alde-Ore Estuary (AOE) SPA are presented in full in Table 7.4 (Appendix 2 – Cumulative and in-combination assessment tables) and summarised below in Table 3.14.

79. The breeding season SPA apportioned figures for other wind farms included in the assessment were estimated using the SNH apportioning method[2]. This uses the distance between the wind farm and each colony in the region and the colony population sizes to calculate the proportion on the wind farm expected to originate from the Alde-Ore Estuary SPA (see Table 7.3 for values used in the calculation).

Table 3.14 Summary lesser black-backed gull in-combination collisions apportioned to the AOE SPA.

Wind farm in-combination scenario*	Breeding season	Nonbreeding season	Annual
Total (all projects)	41.4	15.6	57.0

* Note that no lesser black-backed gulls were apportioned to this SPA from Hornsea Projects Three and Four, therefore only a single total is presented here.

80. Given that tracking studies have revealed low connectivity for the Alde-Ore SPA population with the Norfolk Boreas site (Thaxter et al. 2012, 2015), it is questionable both whether the proposed Norfolk Boreas project would contribute to an in-combination total during the breeding season, and also if any of the wind farms within 141km should be considered. However, as a precautionary assessment with respect to the Alde-Ore SPA population, wind farms within 141km of the Alde-Ore SPA have been considered during the breeding season, on the grounds that only

[2] https://www.nature.scot/sites/default/files/2018-11/Guidance%20-%20Apportioning%20impacts%20from%20marine%20renewable%20developments%20to%20breeding%20seabird%20populations%20in%20SPAs_0.pdf

these wind farms have the potential to contribute to mortality on the SPA population at this time of year. Hence the apportioned breeding season mortality has been included for Greater Gabbard, Gunfleet Sands, Kentish Flats, Kentish Flats Extension, London Array, Scroby Sands, Sheringham Shoal, Thanet, Thanet Extension, Dudgeon, East Anglia ONE, Galloper, East Anglia THREE, Norfolk Vanguard, East Anglia TWO, East Anglia ONE North and Norfolk Boreas. The total breeding season mortality for these wind farms is 41.4 birds (although, it is more likely that the breeding season total should be based on wind farms within the mean foraging range of 72km (Greater Gabbard, East Anglia ONE, East Anglia TWO, East Anglia ONE North, Galloper, London Array) which indicate a total breeding season mortality estimate of 30 collisions).

81. As discussed above, given the large geographical area from which lesser black-backed gulls migrating through the Norfolk Boreas site originate, it is only possible to apportion nonbreeding season mortality to the Alde-Ore SPA population on the basis of its size relative to the wider lesser black-backed gull population. Across all age classes the Alde-Ore Estuary SPA represents approximately 3.3% of the BDMPS autumn population, about 3.3% of the BDMPS spring population and a maximum of 5% of the BDMPS winter population. As noted above, for many wind farms there is insufficient information to determine in which months nonbreeding season collisions occur. Therefore, on the basis of the whole period a weighted Alde-Ore Estuary SPA percentage of 4% has been calculated (5 months at 3.3% and 4 months at 5%). This indicates that up to 15.6 birds (391 x 4%) could die from the Alde-Ore Estuary SPA population during the nonbreeding season.

82. The annual mortality of lesser black-backed gulls from the Alde-Ore SPA is therefore 15.6 during the nonbreeding season and 41.4 during the breeding season 57 in total.

83. Natural mortality for the SPA population (assuming approximately 4,000 adults) would be around 460 individuals at an average adult mortality rate of 11.5% (Horswill and Robinson 2015). In-combination mortality of up to 57 birds attributable to the Alde-Ore SPA population would represent an increase in mortality rate of 12%.

84. As noted above (section 3) these estimates include several sources of precaution, which inflate these collision estimates by up to 2.5 times. Just accounting for built vs. consented wind farm designs reduces collision estimates to around 60% of the unadjusted values (Trinder 2017). In this case that would reduce the in-combination mortality prediction to around 34, equating to an increase in background mortality of 7.4%.

85. A population model has been developed to provide further interpretation of these potential in-combination impacts (MacArthur Green 2019b). This model follows

current NE guidance, utilising a matched-run approach to generate counterfactuals of population size and counterfactuals of population growth rate and was run for a simulated period of 30 years. Summary results are provided in Table 3.15.

Table 3.15. Lesser black-backed gull Alde Ore Estuary SPA population modelling results (see MacArthur Green 2019 for details).

Model	Adult mortality	Counterfactual metric (after 30 years)	
		Growth rate	Population size
Density independent	35	0.991	0.775
	55	0.986	0.669
Density dependent	35	0.998	0.926
	55	0.997	0.881

86. As noted above (paragraph 73), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

87. Taking the modelled adult mortality of 55 (as the worst case), the population growth rate was predicted to be 1.4% lower (0.986) than the baseline using the precautionary density independent model, and 0.3% lower (0.997) using the density dependent model. At the lower modelled adult mortality of 35, the reduction in growth rate was 0.9% (0.991) for the density independent model and 0.2% (0.998) for the density dependent model.

88. Even with the most precautionary combination of methods (see section 3 and use of the density independent model) these reductions in growth rate are small (no more than 1.4%) and therefore are not considered likely to result in a population decline. The more realistic collision estimates, accounting for the reduced impacts from built wind farms compared with the consented designs and more appropriate levels of nocturnal activity, predict a growth rate reduction of no more than 0.3% (density independent) or 0.2% (density dependent), which further reduces any concerns about the impact on the Alde-Ore Estuary SPA population.

89. It is also worth noting that the in-combination collision total predicted for the consented Galloper Wind Farm was estimated to be 85 (at a 99.5% avoidance rate), which the current total (of up to 57) remains well below, despite the addition of several more wind farms to the total. As noted above (paragraph 77), the SPA population is much smaller than the designated size (and has been since 2001), apparently as a consequence of various factors including changes in farming practice in the region and a shift of rural gull populations to urban locations. Thus, while there is a risk that the in-combination mortality will delay the time for the population to achieve the stated aim (to restore the population to 14,000 pairs), the actual contribution to this delay resulting from the in-combination mortality will be

very small and it is other factors which have reduced the population and are likely to continue to be the primary drivers in determining the population status.

90. Given the degree of precaution in collision assessments, it is concluded that there will be no adverse effect on the integrity of the Alde Ore Estuary SPA due to in-combination collisions of lesser black-backed gull.

3.4 Herring gull

3.4.1 EIA Cumulative collisions

91. Updated cumulative collisions for herring gull at offshore wind farms in the North Sea and Channel is provided in Table 7.5 (Appendix 2). The update addresses comments provided by Natural England in their Relevant Representation (RR-099), including:

- Estimates for two additional wind farms (Kentish Flats Extension and Methil);
- A review of the figures for all other wind farm sites to ensure consistency with other project assessments (updated values are presented for Kentish Flats , Thanet Extension, Norfolk Vanguard, East Anglia ONE North and East Anglia TWO; the latter updated from Preliminary Environmental Impact Report to final submission);
- Presentation of collision totals for the following scenarios:
    - With all wind farms (i.e. including all projects at PEIR stage or later);
    - Without the values for the Hornsea Project Three wind farm (as advised by Natural England);
    - Without the values for the Hornsea Project Four wind farm (which has not submitted a final assessment and is therefore currently at PEIR stage); and
    - Without both the Hornsea Project Three wind farm and the Hornsea Project Four wind farm.

92. The total collisions for the four scenarios described above are summarised in Table 3.16.

Table 3.16 Summary herring gull cumulative collisions.

Wind farm cumulative scenario	Breeding season	Nonbreeding season	Annual
Total (all projects)	387.1	425.4	812.5
Total (minus Hornsea Project Three)	386.1	417.1	803.2
Total (minus Hornsea Project Four)	385.3	424.6	809.9
Total (minus Hornsea Projects Three and Four)	384.3	416.3	800.6

93. The cumulative annual total collisions, estimated with a high degree of precaution (section 3) including all projects is 812, which reduces to 803 with Hornsea Project Three omitted and reduces further to 800 with the additional omission of Hornsea Project Four.

94. The largest herring gull BDMPS is 466,511 and the biogeographic population is 1,098,000. The background mortality for these populations, calculated using an all age class mortality rate of 0.172 (see Table 13.13 in APP 226 for details of how this was calculated) are 80,240 and 188,856 respectively. Addition of the precautionary estimates of 800 to 812 to these increases the background mortality by 0.99% to 1.0% (BDMPS) and 0.4% (biogeographic). IUCN class herring gull at the lowest level conservation status (least concern), due to a very large range and an extremely large population size.

95. Thus, assessed against both the BDMPS and the biogeographic population this level of cumulative mortality is below the threshold of detectability and consequently, even with the precautionary assumptions made in this assessment (see section 3), the impact on the herring gull population due to cumulative collisions is considered to be of low magnitude, so for this low sensitivity species the impact significance is minor adverse.

3.5 Great black-backed gull

3.5.1 EIA Cumulative collisions

96. Updated cumulative collisions for great black-backed gull at offshore wind farms in the North Sea and Channel is provided in Table 7.6 (Appendix 2 – Cumulative and in-combination assessment tables). The update addresses comments provided by Natural England in their Relevant Representation (RR-099) including:

- Estimates for two additional wind farms (Kentish Flats Extension and Methil);
- A review of the figures for all other wind farm sites to ensure consistency with other project assessments (updated values are presented for Thanet Extension, Norfolk Vanguard, East Anglia TWO and East Anglia ONE North; the latter two updated from Preliminary Environmental Impact Report to final submission);
- Presentation of collision totals for the following scenarios:
    - With all wind farms (i.e. including all projects at PEIR stage or later);
    - Without the values for the Hornsea Project Three wind farm (as advised by Natural England);
    - Without the values for the Hornsea Project Four wind farm (which has not submitted a final assessment and is therefore currently at PEIR stage); and
    - Without both the Hornsea Project Three wind farm and the Hornsea Project Four wind farm.

97. The total collisions for these scenarios are summarised in Table 3.17.

Table 3.17 Summary great black-backed gull cumulative collisions.

Wind farm cumulative scenarios	Breeding season	Nonbreeding season	Annual
Total (all projects)	209.8	937.5	1144.2
Total (minus Hornsea Project Three)	190.4	890.9	1078.2
Total (minus Hornsea Project Four)	206.8	923.9	1130.6
Total (minus Hornsea Projects Three and Four)	187.4	877.3	1064.6

98. The cumulative annual total collisions, estimated with a high degree of precaution (section 3) including all projects is 1,144, which reduces to 1,078 with Hornsea Project Three omitted and reduces further to 1,065 with the additional omission of Hornsea Project Four.

99. The largest great black-backed gull BDMPS is 91,399 and the biogeographic population is 235,000. The background mortality for these populations, calculated using an all age class mortality rate of 0.185 (see Table 13.13 in APP 226 for details of how this was calculated) are 16,909 and 43,475 respectively. Addition of 1,064 to 1,147 to these increases the background mortality by 6.3% to 6.7% (BDMPS) and 2.4% to 2.6% (biogeographic).

100. Thus, at these levels of cumulative mortality the increase in background mortality indicates that further consideration is necessary. However, for several reasons (listed in Section 3) the cumulative total is considered to be over precautionary with the consequence that the impact magnitude is inflated.

101. It is clear therefore that the current cumulative collision total of 1,144 is highly precautionary and that with adjustments as noted above (section 3) made, the estimated increase in background mortality would be much lower. However, at Natural England's request further consideration is provided using an update to the previous population modelling.

102. Natural England requested that the PVA outputs for this species were updated using the recently developed NE PVA Tool (https://github.com/naturalengland/Seabird_PVA_Tool). The model settings are provided in Section 8.3 and the results from the PVA are presented in Table 3.18.

Table 3.18 Great black-backed gull BDMPS and biogeographic population modelling results using the NE PVA tool.

Population scale	Adult mortality	Density independent counterfactual metric (after 30 years)		Density dependent counterfactual metric (after 30 years)	
		Growth rate	Population size	Growth rate	Population size
BDMPS	1000	0.9882	0.6930	0.9905	0.7446
	1100	0.9870	0.6677	0.9896	0.7225
	1200	0.9859	0.6437	0.9886	0.7014
Biogeographic	1000	0.9954	0.8674	0.9964	0.8944
	1100	0.9950	0.8552	0.9960	0.8845
	1200	0.9945	0.8432	0.9957	0.8746

103. Although both counterfactual measures (of population size and population growth rate) are presented in Table 3.18, the Applicant considers that the counterfactuals of population growth rate are more informative and credible for assessment purposes for the following reasons:

- The counterfactual of population growth rate can be compared to recent and longer-term population trends and represents a measure of the population's resilience and ability to regenerate. It is also relatively insensitive to the absolute value for the baseline rate of growth or direction (positive or negative);
- In contrast the counterfactual of population size is much more sensitive to the predicted population trend (strength of growth and direction). This is particularly true in the absence of density dependence. For example, a population with a positive growth rate will grow exponentially, with the result that very large differences can be obtained in the baseline (unimpacted) population size and impacted population size (neither of which prediction is credible since seabird populations are constrained by factors such as nest site availability, prey availability, etc.).

104. For these reasons the interpretation of PVA outputs focusses on the counterfactuals of population growth rate.

105. The maximum reduction in the BDMPS population growth rate at an adult mortality of 1,200 using the more precautionary density independent model was 1.4% (0.9859). The equivalent reduction for the biogeographic population was 0.55% (0.9945). Using the more realistic density dependent model the maximum reduction in the BDMPS population growth rate was 1.2% (0.9886) and for the biogeographic population was 0.43% (0.9957). IUCN classes great black-backed gull at the lowest

level conservation status (least concern), due to a very large range and an extremely large population size.

106. Thus, the maximum predicted effect was a reduction in the growth rate of 1.4%. The UK great black-backed gull population has remained relatively stable since 1970, with reductions between the approximate 15 year seabird censuses of 7% (1969 to 1985), 4% (1985 to 1998) and 11% (2000 to 2015). Against these natural changes, the maximum change in the growth rate of 1.4%, itself estimated on the basis of several aspects of precaution (e.g. density independent model of the smallest population, no account for built wind farms presenting lower risks than consented designs and over-estimated nocturnal activity rates), is not considered likely to result in any significant, nor detectable effects on the BDMPS or biogeographic populations.

107. In conclusion, the cumulative impact on the great black-backed gull population, assessed against both the BDMPS and the biogeographic populations, resulting from the highly precautionary collision estimates is not at a level which would trigger a risk of population decline but would only result in a slight reduction in the current growth rates. Consequently, the impact on the great black-backed gull population due to cumulative collisions is considered to be of low magnitude, for a species which has low to medium sensitivity and therefore the impact significance is minor adverse.

3.6 Little gull

3.6.1 HRA Project alone

108. Natural England requested that additional assessment be presented giving consideration to uncertainty in the density estimates through the presentation of collision impacts using the upper and lower confidence estimates as well as the mean values.

109. The collision mortality for the Norfolk Boreas site was four individuals with 95% confidence intervals of 0-14, derived from option 2 of the deterministic Band model (Table 13.34 in APP-266). A precautionary estimate of the population size of little gulls visiting the Greater Wash Area of Search is around 10,000 individuals per year, while a more realistic (but still precautionary) estimate is likely to be around 20,000 individuals per year with an upper estimate of 75,000 (Steinen et al. 2007).

110. The Greater Wash SPA designated population of little gull is 1,255, which is 13% of a population of 10,000 or 6.5% of a population of 20,000. On this basis, and assuming collisions would be distributed uniformly throughout the population, this would imply that a mean maximum of 0.5 individuals from the Greater Wash SPA population of little gull could be killed by collisions (13% of 4), or between 0 and 1.8

using the 95% confidence intervals. These figures would be halved on the basis of the more realistic wider population (of 20,000).

111. The only published estimate of little gull survival rates suggests an adult value of 0.8 (Horswill and Robinson 2015), and thus the natural mortality rate is 0.2. At this natural mortality rate, the annual mortality for the wider little gull population will be between 2,000 and 4,000 birds and for the SPA population will be 251 birds. For the SPA population, the threshold of detectability (1%) would therefore be 2.5 mortalities. Even if it is assumed that the wider population is no more than 10,000, using the upper 95% confidence interval mortality for the SPA (1.8) the increase in mortality would be below the 1% threshold for detectability against natural variations.

112. Therefore, even the most precautionary estimate would result in an increase below the threshold at which increases in mortality are considered detectable. Therefore, it is reasonable to conclude that there will be no adverse effect on the integrity of the Greater Wash SPA as a result of little gull collisions at the proposed Norfolk Boreas project alone.

3.6.2 HRA In-combination

113. Given the extremely small potential impact on little gull due to collisions at Norfolk Boreas it is apparent that the likelihood of the project contributing to an in-combination impact is extremely small. Nonetheless, at the request of Natural England (during discussion on the Statement of Common Ground), the mortality figures for little gull at other wind farms with potential connectivity to the Greater Wash SPA population were reviewed (Table 3.19). Note that not all wind farms with potential connectivity to this population have presented collision estimates for little gull (e.g. Dudgeon) and therefore this table only includes those for which collisions were reported.

Table 3.19 Assessed collision rates for little gull at offshore wind farm sites with potential connectivity to the Greater Wash SPA.

Wind farm	Annual collisions	Avoidance rate (%)	Assessed wind farm size	Collisions updated for 99.2% avoidance rate	Built or proposed wind farm size	Collisions updated for built or proposed wind farm
Triton Knoll	65	98	288 x 3.6MW	26	90 x 9.5MW	c. 15
Race Bank	52	98	206 x 3MW	21	91 x 6MW	12
Sheringham Shoal	8	98	108 x 3MW	3	88 x 3.6MW	3
Hornsea Project One	10	98	332 x 3.6MW	4	174 x 7MW	2

Wind farm	Annual collisions	Avoidance rate (%)	Assessed wind farm size	Collisions updated for 99.2% avoidance rate	Built or proposed wind farm size	Collisions updated for built or proposed wind farm
Hornsea Project Two	1.3	98	360 x 5MW	0.5	N/A	0.5
Hornsea Project Three	0.5	99.2	300 x 6MW	0.5	N/A	0.5
Norfolk Vanguard	5.1	99.2	180 x 10MW	5.1	N/A	5.1
Norfolk Boreas	3.9	99.2	180 x 10MW	3.9	N/A	3.9
Total				**64**		**42**

114. Given a regional little gull population of between 10,000 and 20,000 the cumulative total figure (64) represents an increase in background mortality of between 1.6% and 3.2% (although as noted above the population may be as large as 75,000, further reducing the magnitude of potential impact, to an increase in mortality of less than 0.5%).

115. The Greater Wash SPA designated population of little gull is 1,255, which is 12.6% of a population of 10,000 or 6.3% of a population of 20,000. On this basis, and assuming collisions would be distributed uniformly throughout the population, this would imply that a maximum of 8.1 individuals from the Greater Wash SPA population would be at risk of in-combination collisions (12.6% of 64), although using the actual built projects (or planned designs) and noting that Triton Knoll has reduced its capacity to 90 turbines this would reduce to 5.3 individuals.

116. Furthermore, the in-combination collisions for built designs would be reduced to 2.6 individuals on the basis of the more realistic wider population (of 20,000) or 4 individuals for the consented wind farm designs. These would give rise to increases in mortality for the SPA population of between 1.0% (for built projects and the realistic population of 20,000) and 3.2% using the most precautionary combination of consented development predictions and the smallest regional population estimate of 10,000.

117. It should be noted that a very similar total collision estimate of seven individuals was assessed by the Secretary of State (SoS) for the in-combination assessment for the Triton Knoll non-material change application (BEIS 2018). In relation to this estimate the SoS stated:

"*Assuming collisions are attributed evenly amongst the regional population, this equates to 7 individuals from the Greater Wash population. Such a small impact would also be undetectable in the SPA population*."

118. And also:

"*in view of the small impacts quantified above, the Secretary of State considers that an Appropriate Assessment is not required in this case*."

119. Thus, on the basis of either the most precautionary in-combination mortality of 8.1, or a more realistic total of 2.6, the likelihood of an adverse effect on the integrity of the Greater Wash SPA population due to collisions of little gull can be ruled out for the proposed Norfolk Boreas project in-combination with other plans and projects.

4 Displacement assessment

120. Displacement assessments have been produced following Natural England advice (RR-099) and the cumulative displacement assessments presented in the original application (APP-201 and APP-226) have been updated accordingly. However, the Applicant considers that the methods requested by Natural England and used for these updated assessments are over-precautionary and result in greatly over-estimated impacts with highly improbable outcomes. MacArthur Green (2019a and 2019c) presented detailed discussion on over-precaution in offshore wind farm ornithology assessments, and the key aspects with respect to displacement assessments are summarised below:

- **Extent of displacement**. A review of studies conducted at operational wind farms (MacArthur Green 2019c) concluded that an evidence-based, but still precautionary, assessment of displacement of auks by offshore wind farms might assume that their densities would be reduced inside offshore wind farms by 50% relative to densities in the surrounding area, and by 30%, on average, across a 1 km buffer zone surrounding the wind farm. There are very few examples where displacement is greater than this, and many cases where it is much less. This contrasts with Natural England's advice to assess displacement rates of 30% to 70% across the wind farm and a 2km buffer.. Furthermore, almost all published studies have been conducted at relatively old wind farms with relatively small and closely spaced turbines (e.g. turbine spacing at Princes Amalia wind farm is approximately 600m compared with around 1.2km at Hornsea Project One). While the turbines at later wind farms are larger too, the increases in turbine dimensions are an order of magnitude smaller (e.g. 10s of metres) than the increases in spacing between them (e.g. several 100s of metres). It therefore seems very probable that the increase in space between turbines will have a much greater effect in reducing displacement rates than will any opposite effect due to the comparatively much smaller increase in turbine dimensions. And just such an effect was reported by Leopold et al. (2012). Thus, it is reasonable to assume that displacement rates based on studies at older wind farms are very likely to over-estimate the effect at future ones.

- **Mortality resulting from displacement**. The consequences of displacement are less well understood than rates of displacement, and Natural England therefore adopts precautionary values for assessment of up to 10% (i.e. 10% of displaced individuals suffer mortality as a direct result). However, MacArthur Green (2019c) considered all sources of information which could be used to inform this aspect and concluded that realistic levels of mortality for displaced birds would be less than 1% for all the species considered (guillemot, razorbill and red-throated diver). To assume a mortality rate of 1% would therefore be in keeping with the evidence and still remain precautionary. Indeed, Natural England has stated that for auks morality is *'likely to be at the low end of the range'* (of 1% to 10%; RR-099).

121. Therefore, the Applicant questions the basis for the requirement to consider displacement of up to 70% (when the evidence indicates that this is more likely to be less than 50%, and even this is based on what are likely to be unrepresentative, small wind farms), extending up to 2 km from the wind farm (when the evidence indicates that this should be no more than 1 km) and resulting in up to 10% mortality for auks when the evidence indicates that 1% is a more realistic and still precautionary value (and 10% is acknowledged to be unlikely by Natural England).

4.1 Red-throated diver

122. The following sections provide an update to the displacement assessments in the original application (APP-201 and APP-226) as requested by Natural England (RR-099). For the project alone further consideration of uncertainty in the abundance estimates was requested through the presentation of assessment using the 95% abundance confidence intervals (section 4.1.1). For the cumulative and in-combination assessment, the update provides a ‘like-for-like’ assessment using the SeaMast density data, to permit a comparison of impacts across wind farms (Section 4.1.2).

4.1.1 Project alone

123. Natural England requested that additional assessment is presented giving consideration to uncertainty in the abundance estimates through the presentation of displacement impacts using the upper and lower 95% confidence estimates as well as the mean values. Displacement has been presented using the Natural England advised rates of 100% displacement within the wind farm and 4km buffer and 1% to 10% mortality. In addition a 90% displacement rate is presented in combination with

a 1% mortality rate, which provides precautionary evidence-based rates for assessment (see MacArthur Green 2019d for details). Displacement is presented for each season separately and as an annual total in Table 4.1.

Table 4.1 Red-throated diver abundance and displacement estimates at Norfolk Boreas including upper and lower 95% confidence intervals.

Season	Value	Total population at risk of displacement	Total impact, displacement & mortality rates: 90% - 1%	100% - 10%
Autumn	Lower 95% c.i.	0	0.0	0.0
	Mean	23	0.2	2.3
	Upper 95% c.i.	58	0.5	5.8
Nonbreeding	Lower 95% c.i.	69	0.6	6.9
	Mean	156	1.4	15.6
	Upper 95% c.i.	253	2.3	25.3
Spring	Lower 95% c.i.	0	0.0	0.0
	Mean	620	5.6	62.0
	Upper 95% c.i.	1412	12.7	141.2
Annual	Lower 95% c.i.	69	0.6	6.9
	Mean	799	7.2	79.9
	Upper 95% c.i.	1723	15.5	172.3

4.1.1.1.1 Autumn migration, project alone EIA

124. Using the seasonal peak autumn migration abundance on the Norfolk Boreas site (and 4km buffer) of 23 (mean), the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated to be between 0.2 and 2.3 individuals (Table 4.1). Using the upper 95% confidence estimate of population size (58) the mortality was estimated as 0.5 to 5.8 and using the lower 95% confidence estimate was estimated to be zero.

125. The BDMPS for red-throated diver in autumn is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (see Table 13.13 in APP 226 for details of how this was calculated) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of a maximum of 2.3 (mean) to this increases the mortality rate by 0.07%. Using the upper 95% estimate (5.8) the increase in mortality would be 0.19% and using the lower 95% estimate (0) there would be no change.

126. Thus, for all these precautionary combinations of parameters the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration season, the magnitude of effect is assessed as negligible. As the species is of high sensitivity to disturbance, the impact significance is minor adverse.

4.1.1.1.2 Midwinter, project alone EIA

127. Using the seasonal peak midwinter abundance on the Norfolk Boreas site (and 4km buffer) of 156 (mean), the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated to be between 1.4 and 15.6 individuals (Table 4.1). Using the upper 95% confidence estimate of population size (253) the mortality was estimated as 2.3 to 25.3 and using the lower 95% confidence estimate (69) the mortality was estimated as 0.6 to 6.9.

128. The BDMPS for red-throated diver in winter is 10,177 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 the number of individuals expected to die is 2,320 (10,177 x 0.228). The addition of a maximum of 15.6 (mean) to this increases the mortality rate by 0.67%. Using the upper 95% estimate (25.3) the increase in mortality would be 1.09% and using the lower 95% estimate (6.9) the increase in mortality would be 0.29%.

129. Thus, for all but the most precautionary combination of parameters the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the midwinter period, the magnitude of effect is assessed as negligible. As the species is of high sensitivity to disturbance, the impact significance is minor adverse.

4.1.1.1.3 Spring migration, project alone EIA

130. Using the seasonal peak spring migration abundance on the Norfolk Boreas site (and 4km buffer) of 620 (mean), the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated to be between 5.6 and 62 individuals (Table 4.1). Using the upper 95% confidence estimate of population size (1,412) the mortality was estimated as 12.7 to 141 and using the lower 95% confidence was estimated to be zero.

131. The BDMPS for red-throated diver in spring is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of a maximum of 62 (mean) to this increases the mortality rate by 2.0%. Using the upper 95% estimate (141) the increase in mortality would be 4.7% and using the lower 95% estimate (0) there would be no change.

132. The high abundance estimate in spring was very strongly influenced by the March 2017 survey, conducted on the 29th and 30th March, from which the estimated abundance on Norfolk Boreas and the 4km buffer was 1,217. Given the late timing of this survey it is considered very likely to have recorded a pulse of passage movement through the region during spring migration, rather than indicating a resident

population (Wernham et al. 2002). Consequently, since individuals recorded during this period will only have been present for a short span of time the assumption of 10% mortality is likely to considerably over-estimate the magnitude of impact. For these reasons, the evidence based 1% mortality rate is considered to be more appropriate for birds recorded in late March.

133. The Applicant understands Natural England's desire to be precautionary in terms of the width of buffer around the wind farm, percentage displaced and percentage mortality caused by displacement. However, the Applicant continues to believe that a mortality rate of over 1%, as a direct consequence of displacement (as recommended by Natural England), is highly improbable given that red-throated divers are regularly displaced on a frequent basis by ships (for example by regular ferry services to/from Scottish islands), and yet the total annual mortality of adult red-throated divers is only about 10% p.a. (the only peer-reviewed research into red-throated diver survival estimated annual adult mortality at only 8% p.a. (Schmutz 2014), but an unpublished preliminary study in Sweden reported in a newsletter suggested annual mortality of slightly over 10% per annum (Hemmingsson and Eriksson 2002)).

134. For example, based on vantage point surveys, focal bird observations and ferry transects, in Orkney in winter, Jarrett et al. (2018) report data suggesting that red-throated divers wintering in Orkney waters are displaced by ferries, fishing boats and other vessels on average probably several times per week and possibly sometimes several times within a day, with 75% of red-throated divers taking off and flying out of the area when approached by a vessel, and 54% doing so when the vessel was still 200-300m away from the bird. The suggestion that displacement by a static object (in this case a turbine) results in similar (or even higher) levels of disturbance than due to escape activity to avoid a moving vessel is not considered realistic. And furthermore, that this results in an increase in mortality of as much as 1% for every individual displacement event, does not correspond with the empirical evidence of frequent displacement of red-throated divers and their total annual mortality, despite this regular displacement, being only around 10% per annum (and such effects due to vessel movements will have been occurring for several decades at levels similar to those observed). The Applicant therefore considers Natural England's assumption that between 1% and 10% of displaced individuals will suffer mortality to be extremely precautionary, even at the lower end of that precautionary range.

135. Using the evidence based 90% displacement and 1% mortality rate (which remains precautionary, MacArthur Green 2019d), the addition of a maximum of 12.7 (upper 95%) would increase the mortality rate by 0.4%. As outlined above this is considered a more appropriate figure to use for this assessment.

136. Thus, for all these precautionary combinations of parameters the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration season, the magnitude of effect is assessed as negligible. As the species is of high sensitivity to disturbance, the impact significance is minor adverse.

4.1.1.1.4 Summed annual, project alone EIA

137. The summed Norfolk Boreas site (and 4km buffer) displacement mortality for autumn, winter and spring, using the highly precautionary combinations of 90% to 100% displacement and 1% to 10% mortality, is estimated to be between 7 and 80 (mean), 15 and 172 (upper 95%) and 0.6 to 6.9 (lower 95%), although these figures include an unknown degree of double counting due to overlaps in the populations in each period. The majority of this predicted displacement impact (77%) occurs in spring which, as discussed above, is a period when individuals will be passing rapidly through the area. Consequently, the likelihood of mortality due to displacement in spring is very low and the 1% evidence based mortality rate is considered much more appropriate (MacArthur Green 2019d). Within this range, the more realistic (and still precautionary, see above) 1% mortality rate with 100% displacement the annual total would be 8 individuals (with a 95% range of <1 to 17).

138. Using the largest BDMPS population (13,277, Furness, 2015), the number of individuals expected to die is 3,027 (13,277 x 0.228) and using the biogeographic population (27,000, Furness 2015) the number of individuals expected to die is 6,156. The addition of a maximum of 17 (mean) to this increases the mortality rate for the BDMPS population by 0.5% and of the biogeographic population by 0.27%.

139. Thus, on the basis of this highly precautionary assessment the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during all seasons combined, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.1.2 Cumulative assessment using a 'like-for-like' approach

140. Following the review of the Applicant's red-throated diver assessment in the original submission, Natural England advised the Applicant to undertake a 'like for like' cumulative assessment similar to that undertaken for the Thanet Extension wind farm (Vattenfall 2019). This method uses an independent dataset, developed by Bradbury et al. (2014), to estimate the relative contributions from all wind farms in the red-throated diver BDMPS region (South west North Sea, Furness 2015).

141. This approach is necessary due to the variations in how data have been presented in many of the project assessments, which means it is not possible to simply collate the

individual project level impacts, as is typical for a cumulative assessment. Natural England's advice on this aspect was to estimate the abundance of red-throated diver in all the wind farms in the cumulative assessment using the SeaMaST spatial dataset (Bradbury et al. 2014), thereby ensuring that the relative contribution from each project was maintained. The dataset used for this analysis was the nonbreeding combined boat and aerial dataset (BDMPS Non-Breeding Boat Plus Aerial D), which provided the estimated non-breeding season densities (sitting and flying birds summed) using visual aerial survey data collected between 2001 - 2011, and JNCC European Seabirds At Sea (ESAS) boat-based survey data collected between 1979 - 2011.

142. The full details of the methods are included in Vattenfall (2019). In summary the approach was as follows:

- The GIS data source was the SeaMaST 3x3km grid of density estimates for the North Sea produced by Bradbury et al. (2014);
- Wind farm boundaries for projects in the south-west North Sea were overlaid on the density data to obtain consistent abundance estimates for each site;
- The number of red-throated divers potentially displaced by wind farms in the assessment were obtained as the summed totals by development phase (i.e. tiers), defined as operational (1), under construction (2), consented but not constructed (3) and submitted but not determined (4);
- As well as the summed number of birds at risk of displacement in each tier, the percentage of the total in each tier was also calculated; and
- In some cases the 4km buffers for adjacent wind farms overlap. In these cases the earlier site has been prioritised in the calculations and the later site has been clipped to remove the overlap in order to avoid double counting.

143. It should be noted that the red-throated diver distribution presented in Bradbury et al. (2014) confirmed the coastal nature of this species' wintering distribution, with the consequence that grid cells in the GIS outputs beyond this range had too few observations to permit reliable density estimation. Thus, while the aim of the current assessment was to consider all wind farms in the southern North Sea BDMPS (Furness 2015), the reality is that surveys conducted further offshore recorded few if any red-throated divers. Thus, sites for which the SeaMaST data did not yield density estimates were omitted. These were: Dudgeon, Hornsea Project One, Hornsea Project Two, Hornsea Project Three, Hornsea Project Four, Dogger Bank Creyke Beck A and B, Dogger Bank Teesside A and B (now Sofia) and Triton Knoll.

144. Table 4.2 provides the SeaMaST derived abundance estimate for the wind farms included in the assessment and the percentages of the total these represent, to provide a relative indication of the importance of each location.

Table 4.2 Red-throated diver 'like for like' cumulative abundance estimates obtained using the SeaMaST dataset (Bradbury et al. 2014).

Tier	Wind farm	Abundance			Percentage of total	Percentage of total summed by tier
		Wind farm	4km buffer	Total		
1	Blyth Demonstration	0.0	0.5	0.6	0.0	83.7
1	Greater Gabbard & Galloper	35.4	77.9	113.3	3.5	
1	Gunfleet Sands	54.0	487.2	541.2	16.7	
1	Humber Gateway	0.1	0.7	0.8	0.0	
1	Kentish Flats & Kentish Flats Extension	48.6	343.7	392.3	12.1	
1	London Array	337.4	1165.1	1502.6	46.4	
1	Lincs, Lynn and Inner Dowsing	3.1	18.4	21.5	0.7	
1	Race Bank	0.7	2.7	3.4	0.1	
1	Scroby Sands	9.7	80.0	89.6	2.8	
1	Sheringham Shoal	0.1	0.6	0.7	0.0	
1	Teeside	0.0	0.8	0.9	0.0	
1	Thanet	5.7	34.8	40.5	1.3	
1	Westermost Rough	0.1	0.8	0.9	0.0	
2	East Anglia ONE	5.8	16.1	21.9	0.7	0.7
3	East Anglia THREE	5.9	13.2	19.1	0.6	0.6
4	Norfolk Boreas	2.9	3.5	4.6	0.1	15.1
4	Norfolk Vanguard East*	3.0		4.7	0.1	
4	Norfolk Vanguard West	6.4	13.5	19.9	0.6	
4	Thanet Extension	1.8	59.6	61.4	1.9	
4	East Anglia ONE North	96.6	210.3	306.9	9.5	
4	East Anglia TWO	19.0	71.4	90.4	2.8	
	Total	**636.2**	**2601.1**	**3237.2**	**100**	**100**

* Note that the Norfolk Vanguard East 4km buffer and Norfolk Boreas 4km buffers have been combined since they overlap to a large extent. Also, the overlapping section of the Norfolk Vanguard 4km buffer with East Anglia THREE has been assigned to East Anglia THREE following the tier priorities.

145. The majority of red-throated divers recorded on wind farm sites in the southern North Sea were on existing, operational wind farms (84%). This is not unexpected since red-throated divers preferentially forage in shallower water depths which are typically found closer to shore and this is also where initial offshore wind farm development occurred. It is also informative to note that the tier one wind farms (operational) in Table 4.2, account for 84% of the total diver abundance, but only cover 22% of the total wind farm area, while the tier four projects (in planning), which account for 15% of the diver abundance, cover 61% of the total area. Thus, it

is clear that the density and abundance of divers is disproportionately much higher in the tier one operational wind farms than that the tier four ones, and hence any displacement effects on this species are also disproportionately due to smaller, nearshore, operational wind farms.

146. Norfolk Boreas' contribution to the total abundance (all tiers) was 0.1%, which reflects its location in deeper waters on the outer edge of the observed red-throated diver distribution (Bradbury et al. 2014). Furthermore, given this wind farm covers more than twice the area of any of the others in the assessment it is clear that the relative contribution to the overall potential displacement of red-throated divers is extremely small.

147. The like-for-like assessment presented here has provided further evidence that the overall impact on red-throated divers is mostly accounted for by operational wind farms (e.g. London Array with 46% of the total abundance). In addition, recent surveys have found an apparent increase in the population in the Outer Thames Estuary SPA (within which London Array is located; Irwin et al. 2019), or at the very least no evidence for a decline, therefore it would appear that negative effects on this species due to operational wind farms are less severe than suggested by the precautionary approach advised by Natural England. The above considerations (i.e. wind farm location and relative contributions to red-throated diver abundance) indicate that future wind farm developments are very unlikely to result in any significant change to the population's status.

148. Therefore, the conclusions of the original assessment (APP-226) remain valid: the cumulative displacement of red-throated divers will result in an impact of negligible magnitude for this high sensitivity species resulting in an impact of minor adverse significance.

4.2 Guillemot

149. The following sections provide an update to the displacement assessments as requested by Natural England (RR-099). For the project alone, further consideration of uncertainty was requested through the presentation of assessment using the 95% confidence intervals on abundance (section 4.2.1).

150. For the cumulative and in-combination assessment (section 4.2.2), the update includes:

- Updated cumulative and in-combination tables with the addition of Beatrice Demonstrator, Gunfleet Sands, Kentish Flats, Kentish Flats Extension, Methil, Rampion, Scroby Sands and Hornsea Project Four (PEIR);
- Amended values for Thanet Extension and Hornsea Project Three (as advised by Natural England);

- A review of estimates for all other wind farms to ensure consistency across project assessments (updated values are presented for Greater Gabbard, Galloper, Hornsea Project One, Hornsea Project Two, East Anglia ONE North and East Anglia TWO, the values for the latter two were updated from PEIR to ES).

4.2.1 Project alone

151. Displacement is presented for each season and as an annual total, using the displacement and mortality rates advised by Natural England (30% to 70% displacement and 1% to 10% mortality) and also using the evidence based rates of 50% displacement and 1% mortality (MacArthur Green 2019c). In addition, Natural England requested that additional assessment was presented giving consideration to uncertainty in the density estimates through the presentation of displacement impacts using the upper and lower 95% confidence estimates as well as the mean values, and these are presented in Table 4.3.

Table 4.3 Guillemot abundance and displacement estimates at Norfolk Boreas and apportioned to the FFC SPA including upper and lower 95% confidence intervals.

Season	Value	Total population at risk of displacement	Total impact, displacement & mortality rates:			Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates:		
			30% - 1%	50% - 1%	70%- 10%		30% - 1%	50% - 1%	70%- 10%
Breeding	Lower 95% c.i.	1737	5.2	8.7	121.6	0	0.0	0.0	0.0
	Mean	7767	23.3	38.8	543.7	0	0.0	0.0	0.0
	Upper 95% c.i.	14249	42.7	71.2	997.4	0	0.0	0.0	0.0
Nonbreeding	Lower 95% c.i.	8160	24.5	40.8	571.2	359	1.1	1.8	25.1
	Mean	13777	41.3	68.9	964.4	606	1.8	3.0	42.4
	Upper 95% c.i.	19629	58.9	98.1	1374.0	864	2.6	4.3	60.5
Annual	Lower 95% c.i.	9897	29.7	49.5	692.8	359	1.1	1.8	25.1
	Mean	21544	64.6	107.7	1508.1	606	1.8	3.0	42.4
	Upper 95% c.i.	33878	101.6	169.4	2371.5	864	2.6	4.3	60.5

4.2.1.1.1 Breeding season, project alone EIA

152. There are no breeding colonies for any auk species within foraging range of the Norfolk Boreas site (Thaxter et al. 2012). Therefore, it is reasonable to assume that individuals seen during the breeding season are nonbreeding individuals (e.g. immature birds). Since immature seabirds are known to remain in wintering areas, the number of immature birds in the relevant populations during the breeding season may be estimated as 43% of the total wintering BDMPS population for guillemot and razorbill (Furness, 2015). This gives breeding season populations of nonbreeding individuals of 695,441 guillemots (BDMPS for the UK North Sea and Channel, 1,617,306 x 43%).

153. At the average baseline mortality rate for guillemot of 0.140 (see Table 13.13 in APP 226 for details of how this was calculated) the number of individuals expected to die is 97,362 (695,441 x 0.140). Using the precautionary upper displacement and mortality rates (70% and 10% respectively, section 4), the addition of a maximum of 544 (mean) to this increases the mortality rate by 0.6%. If displacement estimated using the upper 95% abundance (997) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 1.0%. If the lower 95% abundance estimate (122) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 0.1%.

154. Thus, only with the most precautionary combination of parameters (worst case assumptions: mortality estimated with 70% displaced and 10% mortality and upper 95% confidence estimate of abundance) would the increase in mortality approach the level at which the change in background mortality might just be detectable against natural variations, however this would therefore not materially affect the population. Therefore, during the breeding season, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

155. This is the same conclusion reached in the original application (APP-226), even though it has introduced additional elements of precaution through the use of abundance from the upper edge of the probability distribution (i.e. values expected no more than 2.5% of the time). Furthermore, in their Relevant Representation (RR-099) Natural England states, in reference to the 1% to 10% mortality range requested for auk displacement assessment:

'*Natural England agrees that the mortality for auks is likely to be at the low end of the range*'

156. However, unlike NE, the Applicant still remains convinced by the evidence that displacement is unlikely to cause mortality of as much as even 1% of displaced auks. NE stated in RR-099 that:

'*We also noted that the evidence review produced by the Vanguard Applicant (in their auk displacement update submitted at Deadline 1 of the examination) did not provide much support to their assertion that a 1% mortality rate is sufficiently precautionary*.'

157. The Applicant disagrees with this opinion. The review (MacArthur Green 2019c) provided several carefully presented lines of evidence to suggest that it is highly unlikely that displacement of auks increases mortality by as much as 1%. In particular, the fact that displacement would only have a negligible influence on the density of auks at sea throughout the unaffected parts of the North Sea, so that any

density-dependent effect of increased competition for food would be negligible at the North Sea scale of suitable habitat for auk survival. In contrast, the Applicant is not aware of any evidence that displacement would be likely to cause mortality rates in excess of 1% either in auks, or in any comparable scenarios with other similar bird species. The Applicant acknowledges that there is no scientific ’proof’ that auk mortality due to displacement will be less than 1%, but considers there to be a large amount of scientific evidence relevant to this question all of which supports the view that induced mortality of auks caused by displacement is very unlikely to exceed 1%. The Applicant would welcome consideration of any evidence that might suggest otherwise.

4.2.1.1.2 Nonbreeding season, project alone EIA

158. At the average baseline mortality rate for guillemot of 0.140 the number of individuals expected to die is 226,423 (1,617,306 x 0.140). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 964 (mean) to this increases the mortality rate by 0.4%. If displacement estimated using the upper 95% abundance (1,374) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 0.6%. If the lower 95% abundance estimate (571) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.2%.

159. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the nonbreeding season, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.2.1.1.3 Summed annual, project alone EIA

160. At the average baseline mortality rate for guillemot of 0.140 the number of individuals from the largest BDMPS population expected to die across all seasons is 226,423 (1,617,306 x 0.140). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 1,508 (mean) to this increases the mortality rate by 0.67%. If displacement estimated using the upper 95% abundance (2,371) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 1.05%. If the lower 95% abundance estimate (693) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 0.3%.

161. The number of individuals from the biogeographic population expected to die across all seasons is 577,500 (4,125,000 x 0.140). Using the precautionary upper

displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 1508 (mean) to this increases the mortality rate by 0.26%. If the upper 95% abundance estimate (2371) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 0.4%. If the lower 95% abundance estimate (693) is assessed with the upper displacement rate (70 /10) the increase in mortality would be 0.1%.

162. Thus, only with the most precautionary combination of parameters (70% displaced,10% mortality and upper 95% confidence estimate of abundance and smallest population scale) would the increase in mortality approach the level at which the change in background mortality might just be detectable against natural variations, however this would therefore not materially affect the population. As noted above, the estimated impact magnitude is considered to be highly precautionary (section 4) and therefore, during all seasons combined, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse. This is the same conclusion reached in the original application (APP-226), even though it has introduced additional elements of precaution through the use of abundance from the upper edge of the probability distribution (i.e. values expected no more than 2.5% of the time).

4.2.1.1.4 Project alone HRA

163. The Norfolk Boreas displacement mortality was apportioned to the FCC SPA on the basis of no connectivity in the breeding season (as the wind farm is located more than three times the mean maximum foraging range for this species) and an even distribution in the nonbreeding season (on the assumption that the SPA population is evenly distributed within the nonbreeding BDMPS population).

164. The baseline mortality is estimated to be 5,051 (calculated using the adult population 83,214 and the mortality rate of 0.0607, Horswill and Robinson 2015). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 42 (mean) to this increases the mortality rate by 0.8%. If the upper 95% abundance estimate (60) is combined with the upper displacement rate (70 /10) the increase in mortality would be 1.2%. If the lower 95% abundance estimate (25) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.5%. Therefore, it is not possible on this basis to rule out the risk of a likely significant effect due to displacement of guillemot from Norfolk Boreas when estimated using the most precautionary rates of displacement and mortality. However, it is important to note that Natural England (RR-099) stated that mortality rates associated with displacement of auks are '*likely to be at the low end of this range'* [1% to 10%]. Therefore, mortality estimates based on a suggested 10% death rate of displaced auks will greatly overestimate likely impacts, perhaps by

a factor of as much as ten. In this case the threshold for an impact exceeding the 1% threshold of detectability is only reached when the most precautionary abundance estimate (upper 95% confidence value) is combined with the upper displacement rate of 70% and an 8.4% mortality rate (i.e. close to the upper end of the range of 1-10%). Nonetheless, further assessment is provided below.

165. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for mortality levels of 50 and 100 (the nearest values to the project alone mean and upper 95% predictions) are provided in Table 4.4.

Table 4.4 Guillemot FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	50	0.999	0.983	Tables A2_9.1 & A2_9.3
	2		0.999	0.983	Tables A2_11.1 & A2_11.3
Density dependent	1		1.000	0.992	Tables A2_10.1 & A2_10.3
	2		1.000	0.991	Tables A2_12.1 & A2_12.3
Density independent	1	100	0.999	0.966	Tables A2_9.1 & A2_9.3
	2		0.999	0.966	Tables A2_11.1 & A2_11.3
Density dependent	1		1.000	0.983	Tables A2_10.1 & A2_10.3
	2		1.000	0.982	Tables A2_12.1 & A2_12.3

166. Although both counterfactual measures (of population size and population growth rate) are presented in Table 4.4, the Applicant considers that the counterfactuals of population growth rate are more informative and credible for assessment purposes for the following reasons:

- The counterfactual of population growth rate can be compared to recent and longer-term population trends and represents a measure of the population’s resilience and ability to regenerate. It is also relatively insensitive to the absolute value for the baseline rate of growth or direction (positive or negative);
- In contrast the counterfactual of population size is much more sensitive to the predicted population trend (strength of growth and direction). This is

particularly true in the absence of density dependence. For example, a population with a positive growth rate will grow exponentially, with the result that very large differences can be obtained in the baseline (unimpacted) population size and impacted population size (neither of which prediction is credible since seabird populations are constrained by factors such as nest site availability, prey availability, etc.).

167. For these reasons the interpretation of PVA outputs focusses on the counterfactuals of population growth rate.

168. The maximum reduction in the population growth rate, at a precautionary mortality of 100 (which is much higher than the predicted maximum of 60), using the more precautionary density independent model was 0.1% (0.999). On the basis that the observed rate at which this population grew between 2000 and 2008 (3.0%) and between 2008 and 2017 (4.0%) (RSPB unpubl. Report 2017), a reduction of 0.1% to this rate represents a negligible risk for the population.

169. The guillemot breeding numbers at the Flamborough and Filey Coast SPA have shown strong growth over the last 20 years and the population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the guillemot population, subject to natural change.

170. On the basis of the population model outputs the number of predicted project alone guillemot displacement mortalities attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a small reduction in the growth rate currently seen at this colony.

171. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA due to displacement impacts on guillemot from the proposed Norfolk Boreas project alone.

4.2.2 Cumulative and in-combination displacement

172. The complete guillemot cumulative estimates are provided in Appendix 2 – Cumulative and in-combination assessment tables and a summary of totals is provided in Table 4.5 for the following scenarios:

- All wind farms (i.e. including all projects at PEIR stage or later);
- Without the values for the Hornsea Project Three wind farm (as advised by Natural England);
- Without the values for the Hornsea Project Four wind farm (which has not submitted a final assessment and is therefore currently at PEIR stage); and

- Without both the Hornsea Project Three wind farm and the Hornsea Project Four wind farm.

Table 4.5 Summary guillemot cumulative abundance on North Sea wind farms.

Wind farms included	Breeding season	Nonbreeding season	Annual
All North Sea	185878	242936	428814
All North Sea minus Hornsea Project Three	172504	223762	396266
Total (minus Hornsea Project Four)	170633	173381	344014
Total (minus Hornsea Projects Three and Four)	157259	154207	311466

173. The annual total guillemot population at risk of displacement on wind farms in the North Sea, with all projects in planning included, is 428,814. Of this total, over 117,000 is accounted for by the Hornsea Project Three and Four wind farms. With these omitted from the calculation the annual total is reduced to 311,466.

174. The population abundances apportioned to the Flamborough and Filey Coast SPA are provided in Table 4.6.

Table 4.6 Summary guillemot in-combination abundance apportioned to the FFC SPA.

Wind farms included	Breeding season	Nonbreeding season	Annual
All North Sea	32701.6	10689.4	43390.8
All North Sea minus Hornsea Project Three	32701.6	9845.7	42547.1
Total (minus Hornsea Project Four)	17456.6	7629	25085.4
Total (minus Hornsea Projects Three and Four)	17456.6	6785.3	24241.7

175. The annual FFC SPA guillemot population at risk of displacement on wind farms in the North Sea, with all projects in planning included, is 43,391. Of this total, over 19,000 is accounted for by the Hornsea Project Three and Four wind farms. With these omitted from the calculation the annual total is reduced to 24,242 (almost half of the total including Hornsea Project Three and Four).

176. Natural England advise auk displacement assessment using precautionary ranges of displacement between 30% and 70% and mortality between 1% and 10%. These are presented in the table below (Table 4.7).

Table 4.7 Guillemot abundance and displacement mortality estimates summed across all UK North Sea wind farms and apportioned to FFC SPA.

Scenario	Season	Total population at risk of displacement	Total impact, displacement & mortality rates: 30% - 1%	50% - 1%	70%-10%	Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates: 30% - 1%	50% - 1%	70%-10%
All North Sea	Breeding	185878	558	929	13011	32701.6	98	164	2289
	Nonbreeding	242936	729	1215	17006	10689.4	32	53	748
	Annual	428814	1286	2144	30017	43391	130	217	3037
All North Sea minus Hornsea Project Three	Breeding	172504	518	863	12075	32701.6	98	164	2289
	Nonbreeding	223762	671	1119	15663	9845.7	30	49	689
	Annual	396266	1189	1981	27739	42547.3	128	213	2978
All North Sea minus Hornsea Project Four	Breeding	170633	512	853	11944	17456.6	52	87	1222
	Nonbreeding	173381	520	867	12137	7629	23	38	534
	Annual	344014	1032	1720	24081	25085.4	75	125	1756
All North Sea minus Hornsea Project Three and Hornsea Project Four	Breeding	157259	472	786	11008	17456.6	52	87	1222
	Nonbreeding	154207	463	771	10794	6785.3	20	34	475
	Annual	311466	934	1557	21803	24241.7	73	121	1697

4.2.2.1 EIA Cumulative

177. The cumulative total annual mortality across all UK North Sea and Channel wind farms (including Hornsea Projects Three and Four) was estimated to be in the range 1,286 to 30,017, across the range of displacement (30% to 70%) and mortality (1% to 10%) percentages. With Hornsea Project Three omitted these are reduced to between 1,189 and 27,739 and with Hornsea Project Four also omitted these decline further to between 934 and 21,803.

178. Assessed against the largest BDMPS (2,045,078), with a baseline mortality rate of 0.14, the addition of the worst case annual displacement mortality of 30,017 (for all wind farms, including both Hornsea Project Three and Hornsea Project Four), at 70% displaced and 10% mortality, would increase the mortality rate by 10.5%, while assessed against the biogeographic population (4,125,000, Furness 2015) this would result in an increase in mortality of 5.2%. Using the evidence based rates (50% displaced and 1% mortality, Norfolk Vanguard, MacArthur Green 2019c) the annual mortality of 2,144 would increase the mortality rate for the BDMPS population by 0.75% and for the biogeographic population the increase would be 0.37%.

179. Therefore, using the most precautionary rates requested by Natural England (70% displaced and 10% mortality), this suggests that a significant cumulative

displacement impact cannot be ruled out when assessed against the BDMPS population (an increase in mortality of 10.5%). However, it should be noted that the contribution to this total from Norfolk Boreas is 5.1% (0.5% of the increase in background mortality). Furthermore, Natural England (2019a, RR-099) stated that mortality rates associated with displacement of auks are 'likely to be at the low end of this range' [1% to 10%]. Therefore, mortality estimates based on a suggested 10% mortality rate of displaced auks will greatly overestimate likely impacts, perhaps by a factor of as much as ten.

180. Using the evidence-based rates (50% displaced and 1% mortality, MacArthur Green 2019c) this increase in mortality is a maximum of 0.75% which is below the threshold considered detectable (1%). On this basis the displacement mortality would have a negligible magnitude for a medium sensitivity species and result in an impact of minor significance. Furthermore, assessment omitting Hornsea Projects Three and Four reduces the predicted totals by almost 30%, further reducing the concern about this potential impact.

4.2.2.2 HRA In-combination

181. Given the low mortality due to Norfolk Boreas it is clear that the project will also make a small contribution to an in-combination impact. Nonetheless, on the basis of the totals presented in Table 7.8, the combined displacement mortality across the whole year, including all wind farms, was estimated to be in the range 130 to 3,037 individuals (Table 4.7; from 30% displaced and 1% mortality to 70% displaced and 10% mortality). These would increase the baseline mortality rate of the SPA population (83,214 adults) by between 2.6% and 60% (although as noted above section 4.2.1.1.1, this may overestimate impacts by up to a factor of 10). Assessed using the evidence based displacement (50%) and mortality (1%) rates, the mortality of 217 would result in an increase of 4.3%.

182. On this basis, using the worst case approach (70% displacement and 10% mortality, and including Hornsea Project Three and Hornsea Project Four, it may not be possible to rule out an adverse effect on the guillemot population due to in-combination displacement effects, however the contribution from Norfolk Boreas is very small, estimated to comprise less than 1.5%.

183. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018), and these were used for the project alone assessment, however the maximum mortality modelled in MacArthur Green (2018) was 1,600 per year, which was insufficient for the current predicted worst case maximum of 3,037.

184. Therefore, the recently developed NE PVA Tool was used for the in-combination assessment (https://github.com/naturalengland/Seabird_PVA_Tool). The model

settings are provided in Appendix 3 – NE Seabird PVA Input parameters added and the results from the PVA are presented in Table 4.8.

Table 4.8 Guillemot FFC SPA population modelling results using the NE PVA tool.

Population scale	Adult mortality	Density independent counterfactual metric (after 30 years)		Density dependent counterfactual metric (after 30 years)	
		Growth rate	Population size	Growth rate	Population size
FFC SPA	100	0.9986	0.9592	0.9987	0.9611
	200	0.9973	0.9204	0.9974	0.9233
	1700	0.9772	0.4895	0.9781	0.5028
	2900	0.9612	0.2932	0.9626	0.3067
	3050	0.9592	0.2744	0.9606	0.2881

185. As noted above (paragraph 166), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

186. The maximum reduction in the population growth rate at a mortality of 3,050 (for the most precautionary 70% displaced and 10% mortality, including Hornsea Project Three and Hornsea Project Four) and using the more precautionary density independent model, was 4.1% (0.9592). While using the density dependent model the growth rate reduction was 3.9% (0.9606).

187. It should be noted that with Hornsea Project Three and Hornsea Project Four omitted from the total, the maximum reduction in the population growth rate at a mortality of 1,700, estimated using the more precautionary density independent model, was 2.3% (0.9772). While using the density dependent model the growth rate reduction was 2.2% (0.9781).

188. On the basis that the observed rate at which this population grew between 2000 and 2008 (3.0%) and between 2008 and 2017 (4.0%) (RSPB unpubl. Report 2017), predictions based on the highly precautionary displacement mortality (70% displaced and 10% mortality) would result in the population stabilising at its current size, but would not result in a decline. However, given the highly precautionary nature of the methods recommended by Natural England, which at 10% mortality potentially over estimates the impact by a factor of ten (see section 4), and the fact that Natural England have noted that mortality is likely to be at the lower end of the 1% to 10% scale, it is very probable that the effect on the population will be considerably smaller and will not prevent future growth. For example, the evidence-based mortality (using 50% displaced and 1% mortality) for all wind farms was 217. This

level or impact, which it is important to stress remains precautionary, is predicted to reduce the growth rate by 0.3% (0.9973; Table 4.8). Thus, it is clear that on the basis of the evidence based assessment, rather than the highly precautionary one, the impact will be much smaller and would result in no detectable effect on the population growth rate.

189. The guillemot breeding numbers at the Flamborough and Filey Coast SPA have shown strong growth over the last 20 years and the population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the guillemot population, subject to natural change.

190. On the basis of the population model outputs the number of predicted in-combination guillemot displacement mortalities attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline and so would not have an adverse effect on integrity of the SPA.

191. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA due to displacement impacts on guillemot from the proposed Norfolk Boreas project in-combination with other plans and projects.

4.3 Razorbill

192. The following sections provide an update to the displacement assessments as requested by Natural England (RR-099). For the project alone, further consideration of uncertainty was requested through the presentation of assessment using the 95% confidence intervals on abundance (section 4.3.1).

193. For the cumulative and in-combination assessment (section 4.3.2), the update includes:

- Updated cumulative and in-combination tables with the addition of Beatrice Demonstrator, Gunfleet Sands, Kentish Flats, Kentish Flats Extension, Methil, Rampion, Scroby Sands and Hornsea Project Four (PEIR);
- Amended values for Thanet Extension and Hornsea Project Three (as advised by Natural England); and
- A review of estimates for all other wind farms to ensure consistency across project assessments (updated values are presented for Firth of Forth Alpha and Bravo, East Anglia ONE North and East Anglia TWO, the latter two updated from PEIR to ES).

4.3.1 Project alone

194. Displacement is presented for each season and as an annual total, using the displacement and mortality rates advised by Natural England (30% to 70%

displacement and 1% to 10% mortality) and also using the evidence-based rates of 50% displacement and 1% mortality (MacArthur Green 2019c). In addition, Natural England requested that additional assessment be presented giving consideration to uncertainty in the density estimates through the presentation of displacement impacts using the upper and lower confidence estimates as well as the mean values and these are presented in Table 4.9.

Table 4.9 Razorbill abundance and displacement estimates at Norfolk Boreas and apportioned to the FFC SPA including upper and lower 95% confidence intervals.

Season	Value	Total population at risk of displacement	Total impact, displacement & mortality rates: 30% - 1%	50% - 1%	70%- 10%	Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates: 30% - 1%	50% - 1%	70%- 10%
Breeding	Lower 95% c.i.	263	0.8	1.3	18.4	0	0.0	0.0	0.0
	Mean	630	1.9	3.2	44.1	0	0.0	0.0	0.0
	Upper 95% c.i.	1050	3.2	5.3	73.5	0	0.0	0.0	0.0
Autumn	Lower 95% c.i.	111	0.3	0.6	7.8	4	0.0	0.0	0.3
	Mean	263	0.8	1.3	18.4	9	0.0	0.0	0.6
	Upper 95% c.i.	443	1.3	2.2	31.0	15	0.0	0.1	1.1
Nonbreeding	Lower 95% c.i.	527	1.6	2.6	36.9	14	0.0	0.1	1.0
	Mean	1065	3.2	5.3	74.6	29	0.1	0.1	2.0
	Upper 95% c.i.	1677	5.0	8.4	117.4	45	0.1	0.2	3.2
Spring	Lower 95% c.i.	110	0.3	0.6	7.7	4	0.0	0.0	0.3
	Mean	345	1.0	1.7	24.2	12	0.0	0.1	0.8
	Upper 95% c.i.	636	1.9	3.2	44.5	22	0.1	0.1	1.5
Annual	Lower 95% c.i.	1011	3.0	5.1	70.8	22	0.1	0.1	1.5
	Mean	2303	6.9	11.5	161.2	49	0.1	0.2	3.5
	Upper 95% c.i.	3806	11.4	19.0	266.4	82	0.2	0.4	5.7

4.3.1.1.1 Breeding season, project alone EIA

195. There are no breeding colonies for any auk species within foraging range of the Norfolk Boreas site. Therefore, it is reasonable to assume that individuals seen during the breeding season are nonbreeding individuals (e.g. immature birds). Since immature seabirds are known to remain in wintering areas, the number of immature birds in the relevant populations during the breeding season may be estimated as 43% of the total wintering BDMPS population for guillemot and razorbill (Furness, 2015). This gives breeding season populations of nonbreeding individuals of 94,007 razorbills (BDMPS for the UK North Sea and Channel, 218622 x 43%).

196. At the average baseline mortality rate for razorbill of 0.174 (see Table 13.13 in APP 226 for details of how this was calculated) the number of individuals expected to die is 16,357 (94,007 x 0.174). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 44 (mean)

to this increases the mortality rate by 0.27%. If displacement is calculated using the upper 95% abundance (1050) and combined with the upper displacement rate (70 /10), the estimated mortality would be 73 and the increase in mortality would be 0.4%. If the lower 95% abundance estimate (263) is combined with the upper displacement rate (70 /10), the estimated mortality would be 18 and the increase in mortality would be 0.1%.

197. Thus, for all combination of parameters the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the breeding season, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.3.1.1.2 Autumn migration, project alone EIA

198. At the average baseline mortality rate for razorbill of 0.174 the number of individuals expected to die is 102,986 (591,874 x 0.174). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 18 (mean) to this increases the mortality rate by 0.02%. If displacement is calculated using the upper 95% abundance (443) and combined with the upper displacement rate (70 /10), the estimated mortality would be 31 and the increase in mortality would be 0.03%. If the lower 95% abundance estimate (111) is combined with the upper displacement rate (70 /10), the estimated mortality would be 8 and the increase in mortality would be <0.01%.

199. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.3.1.1.3 Midwinter, project alone EIA

200. At the average baseline mortality rate for razorbill of 0.174 the number of individuals expected to die is 38,040 (218,622 x 0.174). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 75 (mean) to this increases the mortality rate by 0.2%. If displacement is calculated using the upper 95% abundance (1677) and combined with the upper displacement rate (70 /10), the estimated mortality would be 117 and the increase in mortality would be 0.3%. If the lower 95% abundance estimate (527) is combined with the upper displacement rate (70 /10), the estimated mortality would be 37 and the increase in mortality would be 0.09%.

201. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the midwinter period, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.3.1.1.4 Spring migration, project alone EIA

202. At the average baseline mortality rate for razorbill of 0.174 the number of individuals expected to die is 102,986 (591,874 x 0.174). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 24 (mean) to this increases the mortality rate by 0.02%. If displacement is calculated using the upper 95% abundance (636) and combined with the upper displacement rate (70 /10), the estimated mortality would be 44 and the increase in mortality would be 0.04%. If the lower 95% abundance estimate (110) is combined with the upper displacement rate (70 /10), the estimated mortality would be 8 and the increase in mortality would be <0.01%.

203. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.3.1.1.5 Summed annual, project alone EIA

204. At the average baseline mortality rate for razorbill of 0.174 the number of individuals from the largest BDMPS population expected to die across all seasons is 102,986. Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 161 (mean) to this increases the mortality rate by 0.15%. If the upper 95% abundance estimate (3806) is combined with the upper displacement rate (70 /10), the estimated mortality would be 266 and the increase in mortality would be 0.26%. If the lower 95% abundance estimate (1011) is combined with the upper displacement rate (70 /10), the estimated mortality would be 71 and the increase in mortality would be 0.06%.

205. The number of individuals from the biogeographic population expected to die across all seasons is 297,018 (1,707,000 x 0.174). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of a maximum of 161 (mean) to this increases the mortality rate by 0.05%. If the upper 95% abundance estimate (3806) is combined with the upper displacement rate (70

/10), the estimated mortality would be 266 and the increase in mortality would be 0.09%. If the lower 95% abundance estimate (1011) is combined with the upper displacement rate (70 /10), the estimated mortality would be 71 and the increase in mortality would be 0.024%.

206. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during all seasons combined, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.3.1.1.6 Project alone HRA

207. Apportioning the Norfolk Boreas displacement mortality to the FFC SPA has been carried out on the basis of no connectivity in the breeding season (as the wind farm is located more than three times further from the SPA than the mean maximum foraging range for this species) and an even distribution in the nonbreeding season (on the assumption that the SPA population is evenly distributed within the nonbreeding BDMPS population). The baseline mortality is estimated to be 2,220 (calculated using the FFC adult population of 21,140 and the adult mortality rate of 0.105, Horswill and Robinson 2015). Using the precautionary upper displacement and mortality rates (70% and 10% respectively), the addition of an annual maximum of 3.5 (mean) to this increases the mortality rate by 0.15%. If the upper 95% abundance estimate of 5.7 birds for the upper displacement rate (70 /10) is used then the increase in mortality would be 0.26%. If the lower 95% abundance estimate (1.5) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.07%. Since even when the most precautionary approach is adopted, all the mortality increases are below the threshold of detectability, displacement of razorbill from Norfolk Boreas would not have an adverse effect on the integrity of the SPA.

4.3.2 Cumulative and in-combination displacement

208. The complete razorbill cumulative estimates are provided in Appendix 2 – Cumulative and in-combination assessment tables and a summary of totals is provided in Table 4.10.

Table 4.10 Summary razorbill cumulative abundance on North Sea wind farms.

Wind farms included	Breeding season	Autumn migration	Nonbreeding season	Spring migration	Annual
All North Sea	32704	41066.1	26211.4	33665.0	133644.0
All North Sea minus Hornsea Project Three	32074	39046.1	21187.4	31911.0	124216.0

Wind farms included	Breeding season	Autumn migration	Nonbreeding season	Spring migration	Annual
Total (minus Hornsea Project Four)	32124	35106.1	25526.4	32304.0	125058.0
Total (minus Hornsea Projects Three and Four)	31494	33086.1	20502.4	30550.0	115630.0

209. The annual total razorbill population at risk of displacement on wind farms in the North Sea, with all projects in planning included, is 133,644. Of this total, over 18,000 is accounted for by the Hornsea Project Three and Four projects. With these omitted from the calculation the annual total is reduced to 115,630.

210. The population abundances apportioned to the Flamborough and Filey Coast SPA are provided in Table 4.11.

Table 4.11 Summary razorbill in-combination abundance apportioned to the FFC SPA.

Wind farms included	Breeding season	Autumn migration	Nonbreeding season	Spring migration	Annual
All North Sea	3848.3	1396.2	708.1	1144.3	7097.7
All North Sea minus Hornsea Project Three	3848.3	1327.5	572.5	1084.7	6833.4
Total (minus Hornsea Project Four)	3268.3	1193.6	689.6	1098.0	6250.0
Total (minus Hornsea Projects Three and Four)	3268.3	1124.9	554.0	1038.4	5985.6

211. The annual FFC SPA razorbill population at risk of displacement on wind farms in the North Sea, with all projects in planning included, is 7,098. Of this total, over 1,100 is accounted for by the Hornsea Project Three and Four wind farms. With these omitted from the calculation the annual total is reduced to 5,986.

212. Natural England advise considering auk displacement assessment using ranges of displacement between 30% and 70% and mortality between 1% and 10%. These are presented in the table below (Table 4.12).

Table 4.12 Razorbill abundance and displacement mortality estimates summed across all UK North Sea wind farms and apportioned to FFC SPA.

Scenario	Season	Total population at risk of displacement	Total impact, displacement & mortality rates:			Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates:		
			30% - 1%	50% - 1%	70%-10%		30% - 1%	50% - 1%	70%-10%
All North Sea	Breeding	32704	98	164	2289	3848.3	12	19	269
	Autumn migration	41066.1	123	205	2875	1396.2	4	7	98
	Nonbreeding	26211.4	79	131	1835	708.1	2	4	50

Scenario	Season	Total population at risk of displacement	Total impact, displacement & mortality rates: 30% - 1%	50% - 1%	70%- 10%	Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates: 30% - 1%	50% - 1%	70%- 10%
	Spring migration	33665.0	101	168	2357	1144.3	3	6	80
	Annual	133644.4	401	668	9355	7097.7	21	35	497
All North Sea minus Hornsea Project Three	Breeding	32074	96	160	2245	3848.3	12	19	269
	Autumn migration	39046.1	117	195	2733	1327.5	4	7	93
	Nonbreeding	21187.4	64	106	1483	572.5	2	3	40
	Spring migration	31911.0	96	160	2234	1084.7	3	5	76
	Annual	124216.0	373	621	8695	6833	20	34	478
All North Sea minus Hornsea Project Four	Breeding	32124	96	161	2249	3268.3	10	16	229
	Autumn migration	35106.1	105	176	2457	1193.6	4	6	84
	Nonbreeding	25526.4	77	128	1787	689.6	2	3	48
	Spring migration	32304.0	97	162	2261	1098	3	5	77
	Annual	125058.0	375	625	8754	6250.0	19	31	438
All North Sea minus Hornsea Project Three and Hornsea Project Four	Breeding	31494	94	157	2205	3268.3	10	16	229
	Autumn migration	33086.1	99	165	2316	1124.9	3	6	79
	Nonbreeding	20502.4	62	103	1435	554	2	3	39
	Spring migration	30550.0	92	153	2139	1038.4	3	5	73
	Annual	115630.0	347	578	8094	5985.6	18	30	419

4.3.2.1 EIA Cumulative

213. The cumulative total annual mortality across all UK North Sea and Channel wind farms (including Hornsea Projects Three and Four) was estimated to be in the range 401 to 9,355, across the range of displacement (30% to 70%) and mortality (1% to 10%) percentages. With Hornsea Project Three omitted these are reduced to 373 to 8,695 and with Hornsea Project Four also omitted these decline further to 347 to 8,094.

214. Assessed against the largest BDMPS (591,874), with a baseline mortality rate of 0.174, the addition of the most precautionary displacement mortality of 9,355 (at 70% displaced and 10% mortality) would increase the mortality rate by 9.1%, while assessed against the biogeographic population (1,707,000, Furness 2015) this would result in an increase in mortality of 3.1%. Natural England (2019a, RR-099) stated that mortality rates associated with displacement of auks are ‘likely to be at the low end of this range’ [1% to 10%]. Therefore, mortality estimates based on a suggested

10% mortality rate of displaced auks will greatly overestimate likely impacts, perhaps by a factor of as much as ten.

215. Using the evidence based rates (50% displaced and 1% mortality, MacArthur Green 2019c) the predicted annual mortality of 668 (with all wind farms included) would increase the mortality rate for the BDMPS population by 0.65% and for the biogeographic population the increase would be 0.22%. These increases are below the threshold of detectability and therefore the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse. Furthermore, assessment omitting Hornsea Project Three and Hornsea Project Four reduces the predicted totals by almost 30%, reducing the concern about this potential impact.

4.3.2.2 HRA In-combination

216. Given the small mortality due to Norfolk Boreas it is clear that the Project will also make a small contribution to an in-combination impact. Nonetheless, on the basis of the totals presented in Appendix 2 – Cumulative and in-combination assessment tables, for all wind farms (including Hornsea Project Three and Four) the combined displacement mortality across the whole year was estimated to be in the range 21 to 497 individuals (Table 4.12, from 30% displaced and 1% mortality to 70% displaced and 10% mortality). These would increase the baseline mortality rate of the SPA population (21,140 adults) by between 0.96 and 22.3% (although as noted above in section 4, this almost certainly overestimates impacts by up to a factor of 10). Assessed using the evidence based displacement and mortality rates, the increase would be 1.6%.

217. With Hornsea Project Three and Four omitted, the combined displacement mortality across the whole year was estimated to be in the range 18 to 419 individuals (Table 4.12, from 30% displaced and 1% mortality to 70% displaced and 10% mortality). These would increase the baseline mortality rate of the SPA population (21,140 adults) by between 0.8% and 18.9% (although as noted above section 4, this almost certainly overestimates impacts by up to a factor of 10). Assessed using the evidence based displacement and mortality rates, the increase would be 1.3%.

218. On this basis, using the worst case approach (70% displacement and 10% mortality and including Hornsea Project Three and Hornsea Project Four) there is potential for an adverse effect on the razorbill population due to in-combination displacement effects, however the contribution from Norfolk Boreas is very small, estimated to comprise less than 0.8% (and also note that this may overestimate impacts by a factor of 10).

219. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of

similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for mortality levels of 50 and 500 (the nearest values to the project alone and in-combination predictions) are provided in Table 4.13.

Table 4.13 Razorbill FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	50	0.998	0.934	Tables A2_13.1 & A2_13.3
	2		0.998	0.933	Tables A2_15.1 & A2_13.3
Density dependent	1		1.000	0.978	Tables A2_14.1 & A2_14.3
	2		0.998	0.949	Tables A2_16.1 & A2_16.3
Density independent	1	500	0.976	0.499	Tables A2_13.1 & A2_13.3
	2		0.976	0.499	Tables A2_15.1 & A2_13.3
Density dependent	1		0.994	0.783	Tables A2_14.1 & A2_14.3
	2		0.981	0.567	Tables A2_16.1 & A2_16.3

220. Although both counterfactual measures (of population size and population growth rate) are presented in Table 4.13, the Applicant considers that the counterfactuals of population growth rate are more informative and credible for assessment purposes for the following reasons:

- The counterfactual of population growth rate can be compared to recent and longer-term population trends and represents a measure of the population’s resilience and ability to regenerate. It is also relatively insensitive to the absolute value for the baseline rate of growth or direction (positive or negative);
- In contrast the counterfactual of population size is much more sensitive to the predicted population trend (strength of growth and direction). This is particularly true in the absence of density dependence. For example, a population with a positive growth rate will grow exponentially, with the result that very large differences can be obtained in the baseline (unimpacted) population size and impacted population size (neither of which prediction is credible since seabird populations are constrained by factors such as nest site availability, prey availability, etc.).

221. For these reasons the interpretation of PVA outputs focusses on the counterfactuals of population growth rate.

222. The maximum reduction in the population growth rate, at a mortality of 50 (which is more than eight times the maximum Norfolk Boreas alone displacement mortality of 5.7, estimated using the worst case displacement and mortality rates and upper confidence intervals), using the more precautionary density independent model was 0.2% (0.998). On the basis of the observed rate at which this population has grown, between 2000 and 2008 (7.2%) and between 2008 and 2017 (7.2%) (RSPB unpubl. Report 2017), a reduction of 0.2% to this rate represents a negligible risk for the population. This conclusion is further supported by the fact that the population in 2017 was estimated to be 20,253 pairs (Aitken et al. 2017), more than double the designated population size (10,570 pairs).

223. The maximum reduction in the population growth rate, at a mortality of 500 (which is the nearest modelled value to the in-combination adult total of 497), using the more precautionary density independent model was 2.4% (0.976). On the basis of the observed rate at which this population has grown, between 2000 and 2008 (7.2%) and between 2008 and 2017 (7.2%) (Aitken et al. 2017), a reduction of 2.4% to this rate, due to the most precautionary displacement predictions, would still permit population growth at over 4.8% per year.

224. The razorbill breeding numbers at the Flamborough and Filey Coast SPA have shown strong growth over the last 20 years and are continuing to increase so the population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the razorbill population, subject to natural change.

225. On the basis of the population model outputs, even the highly precautionary estimate of the predicted in-combination razorbill displacement mortalities attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a small reduction in the growth rate currently seen at this colony, and so would not have an adverse effect on integrity of the SPA.

226. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on razorbill due to the proposed Norfolk Boreas project in-combination with other plans and projects.

4.4 Gannet

227. The following sections provide an update to the displacement assessments as requested by Natural England (RR-099). For the project alone, further consideration

of uncertainty was requested through the presentation of assessment using the 95% confidence intervals on abundance (section 4.3.1).

228. For the cumulative and in-combination assessment (section 4.3.2), the update includes:

- Updated cumulative and in-combination tables with the addition of Beatrice Demonstrator, Gunfleet Sands, Kentish Flats, Kentish Flats Extension, Methil, Rampion, Hornsea Project Four (PEIR values) and Scroby Sands;
- Amended values for Thanet Extension and Hornsea Project Three (as advised by Natural England); and
- A review of estimates for all other wind farms to ensure consistency across project assessments (updated values are presented for East Anglia ONE North and East Anglia TWO, updating these from PEIR to ES).

4.4.1 Project alone

229. Displacement is presented for each season and as an annual total, using the displacement and mortality rates advised by Natural England (60% to 80% displacement and 1% mortality). In addition, Natural England requested that additional assessment be presented giving consideration to uncertainty in the density estimates through the presentation of displacement impacts using the upper and lower confidence estimates as well as the mean values. These are presented in Table 4.14.

Table 4.14 Gannet abundance and displacement estimates at Norfolk Boreas and apportioned to the FFC SPA including upper and lower 95% confidence intervals.

Season	Value	Total population at risk of displacement	Total impact, displacement & mortality rates:		Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates:	
			60% - 1%	80% - 1%		60% - 1%	80% - 1%
Breeding	Lower 95% c.i.	0	0.0	0.0	0	0.0	0.0
	Mean	1229	7.4	9.8	1229	7.4	9.8
	Upper 95% c.i.	2733	16.4	21.9	2733	16.4	21.9
Autumn	Lower 95% c.i.	1363	8.2	10.9	65	0.4	0.5
	Mean	1723	10.3	13.8	83	0.5	0.7
	Upper 95% c.i.	2117	12.7	16.9	102	0.6	0.8
Spring	Lower 95% c.i.	322	1.9	2.6	20	0.1	0.2
	Mean	526	3.2	4.2	33	0.2	0.3
	Upper 95% c.i.	765	4.6	6.1	47	0.3	0.4
Annual	Lower 95% c.i.	1685	10.1	13.5	85	0.5	0.7
	Mean	3478	20.9	27.8	1344	8.1	10.8
	Upper 95% c.i.	5615	33.7	44.9	2882	17.3	23.1

4.4.1.1.1 Breeding season, project alone EIA

230. At the average baseline mortality rate for gannet of 0.191 the number of individuals expected to die is 47,441 (using the smallest BDMPS, 248,385 x 0.191). Using the more precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of a maximum mortality of 9.8 (mean) to this increases the mortality rate by 0.02%. If displacement estimated using the upper 95% abundance (21.9) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.05%. If the lower 95% abundance estimate of zero is used there will be no impact.

231. Thus, for all these precautionary combinations of parameters the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the breeding season, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.4.1.1.2 Autumn migration, project alone EIA

232. At the average baseline mortality rate for gannet of 0.191 the number of individuals expected to die is 87,153 (456,298 x 0.191). Using the more precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of a maximum of 13.8 (mean) to this increases the mortality rate by 0.016%. If displacement estimated using the upper 95% abundance (16.9) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.02%. If the lower 95% abundance estimate (10.9) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.01%.

233. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.4.1.1.3 Spring migration, project alone EIA

234. At the average baseline mortality rate for gannet of 0.191 the number of individuals expected to die is 47,441 (248,385 x 0.191). Using the more precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of a maximum of 4.2 (mean) to this increases the mortality rate by <0.01%. If displacement estimated using the upper 95% abundance (6.1) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.01%. If the

lower 95% abundance estimate (2.6) is combined with the upper displacement rate (70 /10) the increase in mortality would be <0.01%.

235. Thus, for all combinations of parameters, including the highly precautionary combination of upper 95% population and upper displacement rates, the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.4.1.1.4 Summed annual, project alone EIA

236. At the average baseline mortality rate for gannet of 0.140 the number of individuals from the largest BDMPS population expected to die across all seasons is 87,153. Using the more precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of a maximum of 27.8 (mean) to this increases the mortality rate by 0.03%. If displacement estimated using the upper 95% abundance (44.9) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.05%. If the lower 95% abundance estimate (13.5) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.015%.

237. The number of individuals from the biogeographic population expected to die across all seasons is 225,380 (1,180,000 x 0.191). Using the more precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of a maximum of 27.8 (mean) to this increases the mortality rate by 0.01%. If the upper 95% abundance estimate (44.9) is combined with the upper displacement rate (70 /10) the increase in mortality would be 0.02%. If the lower 95% abundance estimate (13.5) is combined with the upper displacement rate (70 /10) the increase in mortality would be <0.01%.

238. Thus, for all these precautionary combinations of parameters the increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during all seasons combined, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is minor adverse.

4.4.1.1.5 Project alone HRA

239. Apportioning the Norfolk Boreas displacement mortality to the FFC SPA on the basis of 100% connectivity in the breeding season and an even distribution in the nonbreeding seasons (on the assumption that the SPA population is evenly distributed within the nonbreeding BDMPS population). The baseline mortality is estimated to be 1,770 for the designated population and 2,143 for the more recent

count (calculated using the adult mortality rate of 0.08, Horswill and Robinson 2015). Using the more precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of an annual maximum of 10.8 (mean) birds to this increases the mortality rate by 0.6%. If the upper 95% abundance estimate (23.1) is combined with the upper displacement rate (80 /1) the increase in mortality would be 1.3%. If the lower 95% abundance estimate (0.7) is combined with the upper displacement rate (80 /1) the increase in mortality would be 0.04%. Therefore, only with the most precautionary combination of parameters is the increase in background mortality slightly above the threshold at which this might be detected. However, despite this very small magnitude of effect, obtained with highly precautionary assumptions, further assessment is provided below.

240. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for a mortality level of 25 (the nearest values to the project alone and in-combination predictions) are provided in Table 4.15.

Table 4.15 Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	25	0.999	0.968	Tables A2_1.1 & A2_1.3
	2		0.999	0.968	Tables A2_3.1 & A2_3.3
Density dependent	1		0.999	0.978	Tables A2_2.1 & A2_2.3
	2		0.999	0.978	Tables A2_4.1 & A2_4.3

241. As noted above (paragraph 13), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

242. The maximum reduction in the population growth rate, at an adult mortality of 25 (calculated with the upper 95% abundance and more precautionary displacement and mortality rates of 80% and 1% respectively and using the more precautionary density independent model) was 0.01% (0.999).

243. This compares to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. A reduction of less than 0.01% in this case will have an undetectable effect on the population growth rate and represents a negligible risk for the population.

244. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

245. On the basis of the population model predictions, and despite the application of precautionary methods, the number of predicted displacement caused mortalities at Norfolk Boreas attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a very slight reduction in the growth rate currently seen at this colony.

246. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on gannet due to displacement from the proposed Norfolk Boreas project.

4.4.2 Cumulative and in-combination displacement

247. The complete gannet cumulative estimates are provided in Appendix 2 – Cumulative and in-combination assessment tables and a summary of totals is provided in Table 4.16.

Table 4.16 Summary gannet cumulative abundance on North Sea wind farms.

Wind farms included	Breeding season	Autumn migration	Spring migration	Annual
All North Sea	22156	22570	6629	51355
All North Sea minus Hornsea Project Three	20953	21076	5530	47559
Total (minus Hornsea Project Four)	20264	21378	5970	47612
Total (minus Hornsea Projects Three and Four)	19061	19884	4871	43816

248. The annual total gannet population at risk of displacement on wind farms in the North Sea, with all projects in planning included, is 51,355. Of this total, over 7,500 is accounted for by the Hornsea Project Three and Hornsea Project Four wind farms. With these omitted from the calculation the annual total is reduced to 43,816.

249. The population abundances apportioned to the Flamborough and Filey Coast SPA are provided in Table 4.17.

Table 4.17 Summary gannet in-combination abundance apportioned to the FFC SPA.

Wind farms included	Breeding season	Autumn migration	Spring migration	Annual
All North Sea	8743	1083.2	410.9	10237.1
All North Sea minus Hornsea Project Three	7540	1011.5	342.8	8894.3
Total (minus Hornsea Project Four)	6851	1026	370	8247
Total (minus Hornsea Projects Three and Four)	5648	954.3	301.9	6904.2

250. The annual FFC SPA gannet population at risk of displacement on wind farms in the North Sea, with all projects in planning included, is 10,237. Of this total, over 3,300 is accounted for by the Hornsea Project Three and Four wind farms. With these omitted from the calculation the annual total is reduced to 6,904.

251. Natural England advise gannet displacement assessment using ranges of displacement between 60% and 80% and mortality of 1%. These are presented for the four cumulative and in-combination scenarios in the table below (Table 4.18).

Table 4.18 Gannet abundance and displacement mortality estimates summed across all UK North Sea wind farms and apportioned to FFC SPA.

Scenario	Season	Total population at risk of displacement	Total impact, displacement & mortality rates:		Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates:	
			60% - 1%	80% - 1%		60% - 1%	80% - 1%
All North Sea	Breeding	22156	133	177	8743	52	70
	Autumn migration	22570	135	181	1083.2	6	9
	Spring migration	6629	40	53	410.9	2	3
	Annual	51355	308	411	10237.1	61	82
All North Sea minus Hornsea Project Three	Breeding	20953	126	168	7540	45	60
	Autumn migration	21076	126	169	1011.5	6	8
	Spring migration	5530	33	44	342.8	2	3
	Annual	47559	285	380	8894.3	53	71
All North Sea minus Hornsea Project Four	Breeding	20264	122	162	6851	41	55
	Autumn migration	21378	128	171	1026	6	8
	Spring migration	5970	36	48	370	2	3
	Annual	47612	286	381	8247	49	66
All North Sea minus Hornsea Project Three	Breeding	19061	114	152	5648	34	45
	Autumn migration	19884	119	159	954.3	6	8
	Spring migration	4871	29	39	301.9	2	2

Scenario	Season	Total population at risk of displacement	Total impact, displacement & mortality rates: 60% - 1%	80% - 1%	Population apportioned to FFC SPA	FFC SPA impact, displacement & mortality rates: 60% - 1%	80% - 1%
and Hornsea Project Four	Annual	43816	263	351	6904.2	41	55

4.4.2.1 EIA Cumulative

252. The cumulative total annual mortality across all UK North Sea and Channel wind farms (including Hornsea Project Three and Four) was estimated to be in the range 308 to 411. With Hornsea Project Three omitted these are reduced to 285 to 380 and with Hornsea Project Four omitted these decline further to 263 to 351.

253. Assessed against the largest BDMPS (456,298), with a baseline mortality rate of 0.191, the addition of the worst case displacement mortality of 411 (at 80% displaced and 1% mortality) would increase the mortality rate by 0.47%, while assessed against the biogeographic population (1,180,000) this would result in an increase in mortality of 0.18%.

254. Therefore, even using the most precautionary rates recommended by Natural England (80% displaced and 1% mortality), there would not be detectable effect on the gannet population due to cumulative displacement from North Sea wind farms. The impact would be of negligible magnitude and minor adverse significance.

4.4.2.2 HRA In-combination

255. There is potential that displacement across all wind farms could lead to an in-combination impact on the SPA population. On the basis of the totals presented in Appendix 2 – Cumulative and in-combination assessment tables, the combined displacement mortality including Hornsea Project Three and Four across the whole year was estimated to be in the range 61 to 82 individuals (using 60% to 80% displacement and 1% mortality, Table 4.18). These would increase the baseline mortality rate of the SPA population (22,122 adults) by between 3.4% and 4.6%. With Hornsea Project Three and Four omitted the total across the whole year was estimated to be in the range 41 to 55 individuals (using 60% to 80% displacement and 1% mortality, Table 4.18). These would increase the baseline mortality rate of the SPA population (22,122 adults) by between 2.3% and 3.1%.

256. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density

dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for mortality levels of 50 and 75 (the nearest values to the in-combination prediction) are provided in Table 4.19.

Table 4.19 Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	50	0.998	0.937	Tables A2_1.1 & A2_1.3
	2		0.998	0.936	Tables A2_3.1 & A2_3.3
Density dependent	1		0.999	0.957	Tables A2_2.1 & A2_2.3
	2		0.999	0.957	Tables A2_4.1 & A2_4.3
Density independent	1	75	0.997	0.906	Tables A2_1.1 & A2_1.3
	2		0.997	0.906	Tables A2_3.1 & A2_3.3
Density dependent	1		0.998	0.936	Tables A2_2.1 & A2_2.3
	2		0.998	0.936	Tables A2_4.1 & A2_4.3

257. As noted above (paragraph 13), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

258. The maximum reduction in the population growth rate, at the more precautionary adult mortality of 75 and using the more precautionary density independent model was 0.03% (0.997).

259. This compares to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. A reduction of less than 0.1%, despite the application of precautionary methods, in this case represents a negligible risk for the population.

260. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

261. On the basis of the population model predictions, even the highly precautionary estimate, of the predicted displacement caused mortalities at Norfolk Boreas in-combination with other North Sea wind farms attributed to the Flamborough & Filey

Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a very slight reduction in the growth rate currently seen at this colony,

262. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on gannet due to displacement impacts for the project in-combination with other plans and projects.

4.4.3 Combined collision risk and displacement

263. Natural England considers that gannet are at risk of both collisions and displacement. It is important to note that, on top of the precaution in the individual collision and displacement assessments, summing these two impacts adds another layer of precaution, since it implies that individuals can both be displaced (and suffer increased mortality as a consequence) and also be at risk of collision mortality. This section provides assessment of this combined potential effect for the project alone, cumulatively and in-combination.

4.4.3.1 EIA Project alone

264. The estimated project alone annual collision estimate was 117.6 (95% c.i. 32.4 to 239.6; Table 7.1 and APP-226 Table 13.34) and the project alone annual displacement estimate at 80% displaced and 1% mortality was 27.8 (95% c.i. 13.5 to 44.9; Table 4.14). Thus the summed project alone combined collision and displacement impact was 145.4 (95% c.i. 45.9 to 284.5).

265. Assessed against the largest BDMPS (456,298), with a baseline mortality rate of 0.191, the addition of the worst case mean collision and displacement mortality of 145.4 would increase the mortality rate by 0.17% (95% c.i. 0.05% to 0.33%) while assessed against the biogeographic population (1,180,000) this would result in equivalent increases in mortality of 0.06% (95% c.i. 0.02% to 0.13%).

266. Therefore, using the most precautionary displacement rates recommended by Natural England (80% displaced and 1% mortality) and combining this under the precautionary assumption that gannets are at risk of both displacement and collisions (the latter of which have been estimated on a highly precautionary basis, see section 3), there would not be detectable effect on the gannet population due to impacts at the Norfolk Boreas wind farm. The impact would be of negligible magnitude and minor adverse significance.

4.4.3.2 HRA Project alone

267. The estimated project alone annual collision estimate apportioned to the FFC SPA was 57.4 (95% c.i. 4.1 to 138.0; Table 7.1 and APP-201 Table 6.11) and the project alone annual displacement estimate at 80% displaced and 1% mortality apportioned to the FFC SPA was 10.8 (95% c.i. 0.7 to 23.1; Table 4.14). Thus the summed project

alone combined collision and displacement impact apportioned to the SPA was 68.2 (95% c.i. 4.8 to 161.1).

268. The baseline mortality of the SPA population is estimated to be 1,770 for the designated population and 2,143 for the more recent count (calculated using the adult mortality rate of 0.08, Horswill and Robinson 2015). Using the precautionary upper displacement and mortality rates (80% and 1% respectively), the addition of a maximum of 68 (mean) to this increases the mortality rate by 3.8% (95% c.i. 0.3% to 9.1%).

269. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for a mortality level of 75 and 175 (the nearest values to the project alone predictions, including allowance for the upper 95% confidence interval estimate of 161) are provided in Table 4.20).

Table 4.20 Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	1	75	0.997	0.906	Tables A2_1.1 & A2_1.3
		175	0.992	0.794	
	2	75	0.997	0.906	Tables A2_3.1 & A2_3.3
		175	0.992	0.795	
Density dependent	1	75	0.998	0.936	Tables A2_2.1 & A2_2.3
		175	0.995	0.854	
	2	75	0.998	0.936	Tables A2_4.1 & A2_4.3
		175	0.995	0.854	

270. As noted above (paragraph 13), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

271. The maximum reduction in the population growth rate, for the more precautionary adult mortality of 175 (to allow for the upper 95% confidence interval of the combined estimate of 161) and using the more precautionary density independent model was 0.8% (0.992). Natural England (2019b) suggested that, if the SPA

population follows a similar pattern of growth to those observed at colonies of a similar age, the observed rate of growth is likely to decrease over the coming decades. Natural England (2019b) does not discuss the reasons for this apparent pattern in other colonies, however it is reasonable to assume that this would occur due to increasing levels of competition for resources, in other words density dependence. On this basis the results from the density dependent PVA are appropriate. These indicate a maximum growth rate reduction of 0.5% (0.995), which is still expected to be much less than the overall rate of population growth during the lifetime of the Norfolk Boreas wind farm and thus still represents a negligible risk to the population.

272. This compares to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. A reduction of less than 1% in this case represents a negligible risk for the population.

273. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

274. On the basis of the population model predictions the number of combined collision and displacement caused mortalities at Norfolk Boreas attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a very slight reduction in the growth rate currently seen at this colony.

275. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on gannet due to combined collision and displacement from the proposed Norfolk Boreas project.

4.4.3.3 EIA Cumulative collisions and displacement

276. The cumulative annual total collisions including all projects is 3,157, which reduces to 3,108 with Hornsea Project Three omitted and reduces further to 3,047 with the additional omission of Hornsea Project Four.

277. The cumulative total annual displacement mortality across all UK North Sea and Channel wind farms (including Hornsea Project Three and Hornsea Project Four) was estimated to be in the range 308 to 411. With Hornsea Project Three omitted these are reduced to 285 to 380 and with Hornsea Project Four also omitted these decline further to 263 to 351.

278. Thus, the combined collision and displacement totals are 3,568 (including all projects), 3,488 (with Hornsea Project Three omitted) and 3,398 (with Hornsea Projects Three and Four omitted) and these areas presented in Table 4.21.

Table 4.21 Summary gannet cumulative collisions and displacement.

Wind farm	Annual mortality	
Collisions		
Total (all projects)	3157.1	
Total (minus Hornsea Project Three)	3108.1	
Total (minus Hornsea Project Four)	3095.8	
Total (minus Hornsea Projects Three and Four)	3046.8	
Displacement	**At 60% displacement and 1% mortality**	**At 80% displacement and 1% mortality**
Total (all projects)	308	411
Total (minus Hornsea Project Three)	285	380
Total (minus Hornsea Project Four)	286	381
Total (minus Hornsea Projects Three and Four)	263	351
Combined collisions and displacement		
Total (all projects)	3465.1	3568.1
Total (minus Hornsea Project Three)	3393.1	3488.1
Total (minus Hornsea Project Four)	3381.8	3476.8
Total (minus Hornsea Projects Three and Four)	3309.8	3397.8

279. The largest gannet BDMPS is 456,298 and the biogeographic population is 1,180,000. The background mortality for these populations, calculated using an all age class mortality rate of 0.191 (see Table 13.13 in APP 226 for details of how this was calculated), are 87,153 and 225,380 respectively. The addition of the most precautionary impacts (inc. 80% displaced and 1% mortality) of 3,398 to 3,568 to these increases the background mortality by 3.9% to 4.1% (BDMPS) and 1.5% to 1.6% (biogeographic).

280. Natural England requested that the PVA outputs for this species were updated using the recently developed NE PVA Tool (https://github.com/naturalengland/Seabird_PVA_Tool). The model settings are provided in Appendix 3 – NE Seabird PVA Input parameters and he results from the PVA are presented in Table 4.22.

Table 4.22 Gannet BDMPS and biogeographic population modelling results using the NE PVA tool .

Population scale	Adult mortality	Density independent counterfactual metric (after 30 years)		Density dependent counterfactual metric (after 30 years)	
		Growth rate	Population size	Growth rate	Population size
BDMPS	3400	0.9913	0.7618	0.9914	0.7641
	3500	0.9910	0.7557	0.9911	0.7578
	3600	0.9907	0.7495	0.9908	0.7517
Biogeographic	3400	0.9966	0.9004	0.9967	0.9014
	3500	0.9965	0.8975	0.9966	0.8986
	3600	0.9964	0.8949	0.9965	0.8958

281. As noted above (paragraph 13), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

282. The maximum reduction in the BDMPS population growth rate at an adult mortality of 3,600, using the more precautionary density independent model, was 0.93% (0.9907). The equivalent reduction for the biogeographic population was 0.36% (0.9964).

283. These compare to the observed growth rate for the Scottish population (which holds the majority of the GB population) of 2.9% at which this population has grown over the last 15 years (Murray et al. 2015). While the UK gannet population is reported to have increased by 34% between 2003/4 and 2015[1] (an annual rate of growth approximately 1.4%). Gannet are classed as ‘least concern’ by the IUCN, the lowest level of concern in the IUCN Red List grading system.

284. Thus, assessed against both the BDMPS and the biogeographic population the number of predicted collision and displacement mortalities at North Sea and Channel wind farms, assessed using the application of highly precautionary methods, is not at a level which would trigger a risk of population decline, but would only result in a slight reduction in the current growth rates. Consequently, the cumulative impact on the gannet population due to combined displacement and collisions is considered to be of low magnitude and the impact significance is minor adverse.

4.4.3.4 HRA In-combination

285. The in-combination gannet collisions and displacement apportioned to the FFC SPA are presented in Table 4.23 (Appendix 2 – Cumulative and in-combination assessment tables).

Table 4.23 Summary gannet in-combination collisions and displacement apportioned to the FFC SPA.

In-combination scenario	Collisions			Displacement (80% displacement and 1% mortality)			Combined
	Breeding season	Autumn migration	Spring migration	Breeding season	Autumn migration	Spring migration	Annual
Total (all projects)	336.5	43.1	23.2	70	9	3	484.8
Total (minus Hornsea Project Three)	310.5	42.6	22.5	60	8	3	446.6
Total (minus Hornsea Project Four)	293.2	42.7	22.7	55	8	3	424.6
Total (minus Hornsea Projects Three and Four)	267.2	42.1	22.0	45	8	2	386.3

286. The in-combination combined annual total collisions and displacement including all projects is 485, which reduces to 447 with Hornsea Project Three omitted and is further reduced to 386 with the additional omission of Hornsea Project Four.

287. The FFC SPA population at designation was 22,122 breeding adults, although it was more recently counted at 26,782 (Aitken et al. 2017). The background mortality for these population sizes, calculated using the adult mortality rate of 0.08, are 1,770 and 2,142 respectively. Addition of highly precautionary estimates of between 386 and 485 to these increases the background mortality by between 22% and 27% (designated) or by between 18% and 23% (2017 count).

288. Outputs from a PVA model for this population were presented for the Hornsea Project Three wind farm (MacArthur Green 2018). This modelling was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for mortality levels of 375 and 500 (the nearest values to the project alone and in-combination predictions) are provided in Table 4.24.

Table 4.24 Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
	1	375	0.983	0.609	Tables A2_1.1 & A2_1.3

Model	Demographic rate set	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
			Growth rate	Population size	
Density independent	2		0.983	0.609	Tables A2_3.1 & A2_3.3
Density dependent	1		0.989	0.703	Tables A2_2.1 & A2_2.3
	2		0.989	0.701	Tables A2_4.1 & A2_4.3
Density independent	1	500	0.977	0.515	Tables A2_1.1 & A2_1.3
	2		0.977	0.515	Tables A2_3.1 & A2_3.3
Density dependent	1		0.985	0.616	Tables A2_2.1 & A2_2.3
	2		0.985	0.615	Tables A2_4.1 & A2_4.3

289. As noted above (paragraph 13), the focus of the PVA interpretation is on the counterfactual of population growth rate as this is considered to provide a more robust and credible measure of population status.

290. The maximum reduction in the population growth rate, at a highly precautionary adult mortality of 500 using the more precautionary density independent model was 2.3% (0.977).

291. This compares to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. Therefore a reduction of no more than 2.3% represents a negligible risk for the population. Natural England (2019b) suggested that, if the SPA population follows a similar pattern of growth to those observed at colonies of a similar age, the observed rate of growth is likely to decrease over the coming decades. Natural England (2019b) does not discuss the reasons for this apparent pattern in other colonies, however it is reasonable to assume that this would occur due to increasing levels of competition for resources, in other words density dependence. On this basis the results from the density dependent PVA are appropriate. These indicate a maximum growth rate reduction of 1.5% (0.985), which is still expected to be much less than the overall rate of population growth during the lifetime of the Norfolk Boreas wind farm and thus still represents a negligible risk to the population.

292. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

293. On the basis of the population model predictions the number of predicted collision and displacement mortalities at Norfolk Boreas in-combination with other North Sea wind farms attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a slight reduction in the growth rate currently seen at this colony.

294. Therefore, it can be concluded that, even with the high degree of precaution in the assessment, there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on gannet due to in-combination collision mortality.

4.5 Common scoter

4.5.1 HRA Project alone

295. In the original assessment (APP-201) the Applicant considered that the risk of a likely significant effect (LSE) on over-wintering common scoter in the Greater Wash SPA due to disturbance and displacement during cable installation could be ruled out and that no further assessment was therefore required. However, Natural England considers that there is potential for an LSE and that additional assessment should be provided.

296. There is potential for disturbance and displacement of non-breeding common scoter resulting from the presence of vessels installing the offshore cables for Norfolk Boreas through the Greater Wash SPA. Figure 5.2 in APP-201 presented a map of the Norfolk Boreas offshore cable corridor and the common scoter distribution using the data presented in Natural England and JNCC (2016). This map clearly indicates that there is no overlap with the species' boundary as identified for the SPA and offshore cable corridor (these indicate the main areas this species would be expected to be present). The estimated density of common scoter within the section of the offshore cable corridor which traverses the Greater Wash SPA was in the range 0.0-0.7 birds/km^2, which was the lowest density band identified in Natural England and JNCC (2016).

297. Cable laying vessels are static for large periods of time and move only short distances as cable installation takes place at very low speeds (no more than 400m per hour). Offshore cable installation activity is also a relatively low noise emitting operation, particularly when compared to activities such as piling. The magnitude of disturbance to common scoter for Norfolk Boreas has been estimated on a 'worst case' basis. This assumes that there would be 100% displacement of birds within a 2 km buffer around the source, in this case from up to two cable laying vessels.

298. The number of common scoter that would potentially be at risk of displacement from the Norfolk Boreas offshore cable corridor during the cable laying process, was calculated as the area within 2 km of two cable laying vessels (25.13 km^2) multiplied

by the density range for this species in the SPA (0-0.7/km^2). This precautionary calculation gives a range of 0-18 individuals at risk of displacement, which could temporarily increase the density in the remaining areas of the SPA by 0.5%.

299. As the vessels move, it has been assumed that displaced birds return and therefore any individual will be subjected to only a brief period of impact. It is considered reasonable to assume that birds will return following passage of the vessel since the cable laying vessels will move at 300-400m per hour if surface laying, 150-300m per hour for ploughing or jetting and 30-80m per hour if trenching; this represents a maximum vessel speed of 7m per minute. For context, a modest tidal flow rate for the region is an order of magnitude higher, in the region of 1m per second (i.e. 60m per minute). The tide would therefore be flowing at least nine times faster than the cable laying vessel. Thus, for the purposes of estimating displacement the vessels can be considered as effectively stationary (i.e. from the perspective of the birds affected which will be moving with the tide). Consequently, it can be assumed that the estimated number displaced represents the total number which could be displaced over the course of a single winter, since the zone of exclusion can be treated as fixed (note that the worst case construction schedule allows for two export cables to be installed, potentially occurring in two nonbreeding periods so this effect could occur in a maximum of two years).

300. Definitive mortality rates associated with displacement for common scoter (or for any other seabird species) are not known and precautionary estimates must be used. For this assessment it has been assumed that Natural England would advise the same range as applied to the assessment of displacement for other species, i.e. 1% to 10%. On this basis, of the maximum of 18 individuals at risk of displacement due to cable installation during each of up to two winters, mortality could be between 0 and 2 individuals per winter. The adult mortality for this species is 0.217 (Horswill and Robinson 2015) and the designated population size is 3,449. The background mortality would therefore be 748, and the addition of a highly precautionary two individuals to this would increase the mortality rate by 0.3%, which is below the threshold at which changes are considered to be detectable.

301. It should also be noted that the duration of cable laying would be expected to be no more than approximately 40 days (calculated for a vessel travelling at the slowest speed, 30 metres per hour, with a 12 hour working day traversing the 15km width of the cable route across the SPA). Furthermore, the period of construction within the SPA may not occur during the nonbreeding period when common scoter are present (October to April). Thus, the assumption of up to 10% mortality for such a short-term potential effect is highly precautionary.

302. Therefore, there is no risk of an adverse effect on the integrity of the Greater Wash SPA as a result of temporary displacement of common scoter by cable construction

vessels for Norfolk Boreas alone, even when this is assessed with a high degree of precaution.

4.5.2 HRA In-combination

303. The Greater Wash SPA contains shipping channels within the site that will continue to be subject to maintenance dredging. There may also be a requirement for capital dredging in association with newly developed and future port developments (Defra 2016). Shipping already affects the distribution of common scoter within the SPA and these birds tend to avoid shipping lanes due to disturbance by boats (Defra 2016). This represents a background established situation following many decades of shipping activity in the area. While any increase in shipping activity will constitute an in-combination impact on divers, the low level of project alone risk and absence of other developments in the vicinity of the Norfolk Boreas offshore cable route indicate that the likelihood of an in-combination disturbance effect is extremely small.

304. Nevertheless, there is potential for the cable installation for Norfolk Boreas through the Greater Wash SPA to overlap with that for Norfolk Vanguard and Hornsea Project Three. The cable corridor for Norfolk Vanguard is the same as that for Norfolk Boreas, therefore the potential impact from this project is identical to that for Norfolk Boreas. However, construction for these two projects is extremely unlikely to occur in the same nonbreeding season and therefore at worst these two wind farms could result in a doubling the same very small magnitude effect (an undetectable increase in natural mortality of 0.3% x 2 = 0.6%) taking place in two years.

305. It is not clear from Hornsea Project Three's construction timelines how likely such an overlap would be, and given that the actual duration of cable installation through the SPA for Norfolk Boreas is likely to be no longer than six weeks, it would seem that the risk of this occurring simultaneously is in fact very small. Furthermore, there was no predicted mortality of common scoter due to cable installation displacement for Hornsea Project Three. Therefore in-combination cable installation for Norfolk Boreas and other wind farm cable construction is the same as that predicted for Norfolk Boreas alone (section 4.5.1).

306. It is therefore reasonable to conclude that there will be no adverse effect on the integrity of the Greater Wash SPA as a result of common scoter displacement due to cable laying for the proposed Norfolk Boreas project in-combination with that for Norfolk Vanguard and Hornsea Project Three.

4.6 Flamborough and Filey Coast SPA – seabird assemblage

4.6.1 HRA Project alone

307. The seabird assemblage of the Flamborough and Filey Coast SPA comprises gannet, fulmar, kittiwake, guillemot, razorbill, puffin, herring gull, shag and cormorant. Four of these species have been assessed as individual named features (gannet, kittiwake, guillemot and razorbill) as discussed above (sections 3.2.2, 4.2.1.1.4, 4.3.1.1.6, 4.4.1.1.5, 4.4.3.2) and it has been concluded that there will be no adverse effect on the integrity on the SPA in relation to these species due to Norfolk Boreas alone.

308. The remaining assemblage species are considered to either have no likelihood of connectivity due to small foraging ranges (Thaxter et al. 2012) or coastal preferences (herring gull, shag and cormorant), not considered to be at risk of impacts at wind farms (fulmar, which flies at very low levels and therefore has negligible collision risk and is not considered to be at risk of displacement due to very wide ranging habits and low densities) or were recorded in such low numbers that there is no risk of an impact on the population (puffin, with observations in February and March only and wind farm plus 2km abundances of 6 and 23 in these months respectively, which gives an apportioned Flamborough and Filey population of <0.1 individual).

309. Therefore, on the basis that there are not considered to be any risks of adverse effect on the integrity of the Flamborough and Filey Coast SPA due to impacts on the individual components of the seabird assemblage feature it can be concluded that there will be no risk of adverse effect on the integrity on the seabird assemblage feature itself.

4.6.2 HRA In-combination

310. Since it has been concluded that Norfolk Boreas will not have in-combination adverse effect on the integrity on any of the individual components of the seabird assemblage feature for which individual assessments have been undertaken (gannet, kittiwake, guillemot and razorbill, see sections 3.1.2, 3.2.3, 4.2.2.2, 4.3.2.2, 4.4.2.2, 4.4.3.4), and the additional species (herring gull, fulmar, puffin, shag and cormorant) are not considered to be at risk of adverse effects (as outlined above) it can therefore be concluded that there will not be an adverse effect on the integrity of the Flamborough and Filey Coast SPA due to an in-combination effect on the seabird assemblage feature.

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6 Appendix 1 – Analysis of kittiwake age ratios from aerial survey data

311. Kittiwakes have a distinct juvenile plumage which is characterised by a thick black bar that runs diagonally across the wings from the trailing edge of the inner secondary coverts onto the outermost four primaries. There is also a characteristic black band across the dorsal side of the neck, and a broad black tail band. This is known as the 'tarrock' plumage. These features allow relatively easy discrimination between juvenile kittiwakes and older kittiwakes when photographed from above.

312. The 'tarrock' plumage is lost at the bird's first moult, which occurs when the bird is one year old (in July of the year after the bird was fledged). At that point the bird transitions into what is known as 'second year' plumage. This is retained for one year, until July when the bird reaches two years old, when the bird then moults into its 'third year' plumage which is almost, but not quite, identical to adult plumage (Coulson 1959). The majority of kittiwakes start to breed when four or five years old (Porter and Coulson 1987), but individual variation is high with birds first breeding between two and eight years old (Coulson 2011). Because of that variability in age at recruitment, the category 'immature' kittiwakes includes some individuals up to at least seven years old, while 'breeding adults' may include a few two year old birds. Therefore, there is considerable overlap of ages of 'immatures' and 'adults'.

313. It is difficult to identify birds that are two to three years old ('third year' birds) from plumage, although there are some details that can be used to identify 90% of these individuals (Coulson 1959) if they can be examined in the hand or from high quality photographs taken with a suitable telephoto lens. However, it is impossible from digital aerial photographs to distinguish between 'adult' kittiwakes and birds that are two to three years old, or between birds that are breeding adults and birds that are still nonbreeding immatures. Therefore, the best we could hope for from digital aerial surveys is that most juvenile kittiwakes can be identified from their 'tarrock' plumage, and that an unknown but probably small proportion of one to two year old kittiwakes may be identified as still retaining some immature plumage. Birds that are two years old or older plus some first year birds will normally be pooled into a category of 'kittiwakes that appear consistent with adult plumage because no details identify them as being in immature plumage'. This will be a mixture, of unknown proportions, of older immature birds and breeding adult birds.

314. It is important to point out that it is impossible to identify adult kittiwakes from digital aerial photographs, because it is impossible to be confident from digital aerial images that these individuals do not have any remaining immature feathers.

315. Coulson (1959) estimated that during the early part of the breeding season about 49% of a UK kittiwake population comprises immature birds, while at the end of the breeding season with the addition of juvenile birds this rises to 61%. Similar

proportions, with about 47% of the population being immatures across the year as a whole, were calculated with more recent demographic data by Furness (2015). Coulson (1959) further estimated that of the 49% which are immatures, over half (58%) are in plumages which in the field are unlikely to be distinguished from breeding adults.

316. Digital aerial survey data are not even adequate in some instances to identify the species of bird. Some birds are categorised as 'small gull species' rather than to species. In the present study, 123 birds could not be identified to species but were classified as 'small gull species'. Of these, ages were not assigned to 120 individuals. Surprisingly, three of these were classified as 'adults' despite the fact that the species could not be identified. If the species cannot be identified it is evidently impossible to be confident of the age of the bird, since criteria for age determination require knowledge of the species before age classification can be made, so this raises doubt about the age classification process followed. The classifications of the plumages of birds identified as kittiwakes are given in Table 6.1.

Table 6.1 Plumage classifications of kittiwakes identified in digital aerial surveys.

Month	Age unknown but not juvenile*	Completely unknown age	Age known: Juvenile (tarrock)	Age known: post-juvenile 1st winter	Age known: post-juvenile 1st summer
1	189	2	5	6	
2	84	31		6	5
3	33	13		5	
4	60	8		1	
5	60	57			
6	26	2		2	6
7	59	74	2		
8	20	14	2		
9	19	10	1		
10	32	46			
11	187	28	6	21	
12	395	131	10	8	
Total	1164	416	26	49	11

* Categorised as 'adult' in the survey data, but as noted above this grouping is likely to include sub-adults.

317. It is clear from Table 1 that for a large proportion of kittiwakes no assessment of age was made; 416 of the birds (25%) were classified as of completely unknown age. With such a high percentage that could not be assigned into any age category, it is difficult to have confidence in the proportions aged from the remaining 75%. Only 26 birds were identified as being in juvenile plumage, whereas 1,224 were assigned age classes older than juvenile. 60 birds were classified as 1 to 2 year olds (‘1st summer’ or ‘1st winter’), but it is impossible to assess what proportion of the birds were 1 to 2 year olds because the majority of individuals were classified as age unknown but older than juvenile (1,164 individuals). An unknown, but possibly high, proportion of these will also be 1 to 2 year old birds. No birds were classified as 2 to 3 years old, so that age class remains completely unidentified from the digital aerial photography.

318. There are other inconsistencies in the data that further suggest that the age classifications are unlikely to be reliable. For example, five birds were classified as being in 1st summer plumage in February, which is unlikely to be the case given that birds would normally be in winter plumage in winter. Two birds were classified as being in 1st winter plumage in June, which is also improbable as birds are normally in summer plumage in summer. The apparent age distributions, and percentage of non-aged birds in the samples, also vary quite widely between months. For example in the May sample not a single bird was classified as juvenile or immature and almost 50% of birds were classed as ‘completely unknown age’. In contrast, in June, only 6% of individuals were classed as ‘completely unknown age’, with 24% classed as juvenile or 1 to 2 years old. While there could be biological reasons for variations in age classes recorded, it is less clear why there would be such wide variations in the non-aged percentage is less (min: 1%, max: 55%).

319. We therefore conclude that the estimated kittiwake ages from the digital aerial survey data are not reliable for estimating the proportions of different age classes of kittiwakes present in the survey area.

7 Appendix 2 – Cumulative and in-combination assessment tables

320. Updated wind farm values compared with those in the original application (APP-201 and APP-226) are highlighted in bold font. In all cases, as far as possible the figures for other wind farms are those obtained using Band (2012) option 1 or 2, however in some cases (e.g. for older wind farms) it has been assumed that the figures were obtained using the basic model. In no cases have extended model (options 3 or 4) figures been knowingly included. Collisions are all presented using the advised species specific avoidance rates (gannet and kittiwake: 98.9%; large gulls: 99.5%). It should also be noted that the collision estimates for Hornsea Project Four (PEIR) were estimates using the Marine Scotland Science stochastic CRM (sCRM) which the Applicant has found to generate incorrect values. However, these are the only values available for this project and it has thus been necessary to include these here.

7.1 Cumulative and In-combination Collision Risk Tables

Table 7.1 Gannet cumulative and in-combination collision risk.

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
1	Beatrice Demonstrator	0.6	0.0	0.9	0.04	0.7	0.05	2.2	0.1
1	Greater Gabbard	14.0	0.0	8.8	0.42	4.8	0.30	27.5	0.7
1	Gunfleet Sands	-	-	-	-	-	-	-	-
1	Kentish Flats	1.4	0.0	0.8	0.04	1.1	0.07	3.3	0.1
1	**Kentish Flats Extension**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**
1	Lincs	2.1	2.1	1.3	0.06	1.7	0.10	5.0	2.3
1	London Array	2.3	0.0	1.4	0.07	1.8	0.11	5.5	0.2
1	Lynn and Inner Dowsing	0.2	0.2	0.1	0.01	0.2	0.01	0.5	0.2
1	Scroby Sands	-	-	-	-	-	-	-	-
1	Sheringham Shoal	14.1	14.1	3.5	0.17	0.0	0.00	17.6	14.3
1	Teesside	4.9	2.4	1.7	0.08	0.0	0.00	6.7	2.5
1	Thanet	1.1	0.0	0.0	0.00	0.0	0.00	1.1	0.0
1	Humber Gateway	1.9	1.9	1.1	0.05	1.5	0.09	4.5	2.0
1	Westermost Rough	0.2	0.2	0.1	0.01	0.2	0.01	0.5	0.2
1	Hywind	5.6	0.0	0.8	0.04	0.8	0.05	7.2	0.1

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
2	Kincardine	3.0	0.0	0.0	0.00	0.0	0.00	3.0	0.0
2	Beatrice	37.4	0.0	48.8	2.34	9.5	0.59	95.7	2.9
2	Dudgeon	22.3	22.3	38.9	1.87	19.1	1.18	80.3	25.3
2	Galloper	18.1	0.0	30.9	1.48	12.6	0.78	61.6	2.3
2	Race Bank	33.7	33.7	11.7	0.56	4.1	0.25	49.5	34.5
2	Rampion	36.2	0.0	63.5	3.05	2.1	0.13	101.8	3.2
2	Hornsea Project One	11.5	11.5	32.0	1.54	22.5	1.40	66.0	14.4
3	Blyth Demonstration Project	3.5	0.0	2.1	0.10	2.8	0.17	8.4	0.3
3	**Dogger Bank Creyke Beck Projects A and B**	**66.7**	**33.4**	**68.7**	**3.3**	**44.7**	**2.8**	**180.1**	**39.4**
3	East Anglia ONE	3.4	3.4	131.0	6.29	6.3	0.39	140.7	10.1
3	European Offshore Wind Deployment Centre	4.2	0.0	5.1	0.25	0.1	0.00	9.3	0.3
3	Firth of Forth Alpha and Bravo	800.8	0.0	49.3	2.37	65.8	4.08	915.9	6.4
3	Inch Cape	336.9	0.0	29.2	1.40	5.2	0.32	371.3	1.7
3	**Methil**	**6.0**	**0.0**	**0.0**	**0.00**	**0.0**	**0.00**	**6.0**	**0.0**
3	Moray Firth (EDA)	80.6	0.0	35.4	1.70	8.9	0.55	124.9	2.3
3	Neart na Gaoithe	143.0	0.0	47.0	2.26	23.0	1.43	213.0	3.7
3	Dogger Bank Teesside Projects A and B	14.8	7.4	10.1	0.49	10.8	0.67	35.7	8.5
3	Triton Knoll	26.8	26.8	64.1	3.08	30.1	1.87	121.0	31.7
3	Hornsea Project Two	7.0	7.0	14.0	0.67	6.0	0.37	27.0	8.0
4	East Anglia THREE	6.1	6.1	33.3	1.60	9.6	0.60	49.0	8.3
5	**Hornsea Project Three**	**26.0**	**26.0**	**12.0**	**0.58**	**11.0**	**0.68**	**49.0**	**27.3**

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
5	Thanet Extension	0.0	0.0	11.1	0.53	22.9	1.42	34.0	2.0
5	**Norfolk Vanguard**	**17**	**17.0**	**38**	**1.82**	**11**	**0.68**	**66**	**19.5**
6	Moray West	10.0	0.0	2.0	0.10	1.0	0.06	13.0	0.2
6	Norfolk Boreas	54.1	54.1	48.5	2.33	15.0	0.93	117.6	57.4
6	**East Anglia TWO**	**12.7**	**12.7**	**28.7**	**1.38**	**5.6**	**0.35**	**47.0**	**14.4**
6	**East Anglia ONE North**	**11.0**	**11.0**	**12.8**	**0.61**	**3.4**	**0.21**	**27.2**	**11.8**
6	***Hornsea 4 (PEIR)***	***43.3***	**43.3**	***9.9***	**0.48**	***8.1***	**0.50**	***61.3***	**44.3**
	Total (all projects)	**1884.2**	**336.5**	**898.9**	**43.1**	**373.9**	**23.2**	**3156.9**	**402.8**
	Total (minus Hornsea Project Three)	**1858.2**	**310.5**	**886.9**	**42.6**	**362.9**	**22.5**	**3107.9**	**375.6**
	Total (minus Hornsea Project Four))	**1841.2**	**293.2**	**888.7**	**42.7**	**365.8**	**22.7**	**3095.6**	**358.6**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**1815.2**	**267.2**	**876.7**	**42.1**	**354.9**	**22.0**	**3046.6**	**331.3**

Table 7.2 Kittiwake cumulative and in-combination collision risk.

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
1	Beatrice Demonstrator	0.0	0.0	2.1	0.11	1.7	0.12	3.8	0.2
1	Greater Gabbard	1.1	0.0	15.0	0.81	11.4	0.82	27.5	1.6
1	Gunfleet Sands	-	-	-	-	-	-	-	-
1	Kentish Flats	0.0	0.0	0.9	0.05	0.7	0.05	1.6	0.1
1	**Kentish Flats Extension**	**0.0**	**0.0**	**0.0**	**0.00**	**2.7**	**0.19**	**2.7**	**0.2**
1	Lincs	0.7	0.7	1.2	0.06	0.7	0.05	2.6	0.8
1	London Array	1.4	0.0	2.3	0.12	1.8	0.13	5.5	0.3
1	Lynn and Inner Dowsing	-	-	-	-	-	-	-	-
1	Scroby Sands	-	-	-	-	-	-	-	-
1	Sheringham Shoal	-	-	-	-	-	-	-	-
1	Teesside	38.4	0.0	24.0	1.30	2.5	0.18	64.9	1.5
1	Thanet	0.2	0.0	0.5	0.03	0.4	0.03	1.1	0.1
1	Humber Gateway	1.9	1.9	3.2	0.17	1.9	0.14	7.0	2.2
1	Westermost Rough	0.1	0.1	0.2	0.01	0.1	0.01	0.5	0.1
1	Hywind	16.6	0.0	0.9	0.05	0.9	0.06	18.3	0.1
2	Kincardine	22.0	0.0	9.0	0.49	1.0	0.07	32.0	0.6
2	Beatrice	94.7	0.0	10.7	0.58	39.8	2.87	145.2	3.4
2	Dudgeon	-	-	-	-	-	-	-	-
2	Galloper	6.3	0.0	27.8	1.50	31.8	2.29	65.9	3.8
2	Race Bank	1.9	1.9	23.9	1.29	5.6	0.40	31.4	3.6
2	Rampion	54.4	0.0	37.4	2.02	29.7	2.14	121.5	4.2
2	Hornsea Project One	44.0	36.5	55.9	3.02	20.9	1.50	120.8	41.0
3	Blyth Demonstration Project	1.7	0.0	2.3	0.12	1.4	0.10	5.4	0.2
3	**Dogger Bank Creyke Beck Projects A and B**	**196.7**	**38.0**	**92.0**	**4.97**	**201.3**	**14.5**	**490.0**	**57.4**
3	East Anglia ONE	1.8	0.0	160.4	8.66	46.8	3.37	209.0	12.0
3	European Offshore Wind Deployment Centre	11.8	0.0	5.8	0.31	1.1	0.08	18.7	0.4
3	Firth of Forth Alpha and Bravo	153.1	0.0	313.1	16.91	247.6	17.83	713.8	34.7
3	Inch Cape	13.1	0.0	224.8	12.14	63.5	4.57	301.4	16.7
3	**Methil**	**0.4**	**0.0**	**0.0**	**0.00**	**0.0**	**0.00**	**0.4**	**0.0**
3	Moray Firth (EDA)	43.6	0.0	2.0	0.11	19.3	1.39	64.9	1.5

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
3	Neart na Gaoithe	32.9	0.0	56.1	3.03	4.4	0.32	93.4	3.3
3	Dogger Bank Teesside Projects A and B	136.9	26.4	90.7	4.90	216.9	15.62	444.5	46.9
3	Triton Knoll	24.6	24.6	139.0	7.51	45.4	3.27	209.0	35.4
3	Hornsea Project Two	16.0	13.3	9.0	0.49	3.0	0.22	28.0	14.0
4	East Anglia THREE	6.1	0.0	69.0	3.73	37.6	2.71	112.7	6.4
5	**Hornsea Project Three**	**187.5**	**176.3**	**94.6**	**5.11**	**15.0**	**1.08**	**297.1**	**182.4**
5	Thanet Extension	2.3	0.0	5.3	0.29	15.3	1.10	22.9	1.4
5	**Norfolk Vanguard***	**45.3**	**5.1**	**31.5**	**1.70**	**38.7**	**2.79**	**115.5**	**9.6**
6	Moray West	79.0	0.0	24.0	1.30	7.0	0.50	110.0	1.8
6	Norfolk Boreas	41.7	10.9	113.7	6.1	47.4	3.4	202.8	20.4
6	**East Anglia TWO**	**19.8**	**0**	**9.3**	**0.5**	**20.9**	**1.5**	**50.0**	**2.0**
6	**East Anglia ONE North**	**18.6**	**0**	**12.1**	**0.65**	**27.3**	**1.9**	**58.0**	**2.6**
6	***Hornsea 4 (PEIR)***	***153.3***	***153.3***	***34.7***	***1.87***	***9.9***	***0.71***	***197.9***	**155.9**
	Total (all projects)	**1469.9**	**489.0**	**1704.3**	**92.0**	**1223.4**	**88.1**	**4397.7**	**668.8**
	Total (minus Hornsea Project Three)	**1282.4**	**312.7**	**1609.8**	**86.9**	**1208.4**	**86.9**	**4100.6**	**486.4**
	Total (minus Hornsea Project Four)	**1316.6**	**335.7**	**1669.7**	**90.1**	**1213.5**	**87.3**	**4199.8**	**512.9**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**1129.1**	**159.4**	**1575.1**	**85.0**	**1198.5**	**86.2**	**3902.7**	**330.5**

* Note that the figure for Norfolk Vanguard in the breeding season is based on consideration of different levels of connectivity for Norfolk Vanguard West and Norfolk Vanguard East as discussed in Norfolk Vanguard (2019).

Table 7.3 Lesser black-backed gull Alde-Ore Estuary breeding season apportioning estimates for wind farms included in the in-combination assessment, calculated using the SNH apportioning tool[2]. The colony population sizes (pairs) used were; Great Yarmouth 750, Southtown 450, Lowestoft 2,000, Alde-Ore Estuary ,2000, Felixstowe 700, Ipswich 250 and Outer Trial Bank 1,300.

Wind farm	Distance to lesser black-backed gull colony (km)							Alde-Ore Estuary SPA breeding season proportion
	Great Yarmouth	Southtown	Lowestoft	Alde-Ore Estuary SPA	Felixstowe	Ipswich	Outer Trial Bank	
Greater Gabbard	73	72	59	28	40	50	148	0.65
Gunfleet Sands	101	100	89	46	25	35	143	0.35
Kentish Flats	134	132	119	77	56	64	166	0.38
Kentish Flats Extension	134	132	119	77	56	64	166	0.38
London Array	101	100	88	43	31	45	153	0.46
Scroby Sands	3	4	15	55	80	76	105	0.01
Sheringham Shoal	62	64	76	107	124	115	67	0.15
Thanet	127	126	114	74	64	74	181	0.43
Dudgeon	67	68	80	120	136	130	84	0.15
Galloper	73	72	59	28	40	50	148	0.65
East Anglia ONE	62	60	53	53	77	85	158	0.37
East Anglia THREE	70	70	69	91	118	123	169	0.24
Thanet Extension	127	126	114	74	64	74	181	0.43
Norfolk Vanguard	51	55	60	92	120	120	140	0.17
Norfolk Boreas	75	76	80	111	140	143	170	0.21
East Anglia TWO	43	42	34	34	57	63	150	0.39
East Anglia ONE North	41	40	37	52	78	85	143	0.24

Table 7.4 Lesser black-backed gull cumulative and in-combination collision risk. Breeding season apportioning rates use the values in Table 7.3.

Tier	Wind farm	Breeding season		Nonbreeding season		Annual	
		Total	AOE SPA	Total	AOE SPA	Total	AOE SPA (nonbreeding season apportioned plus breeding season for wind farms <141km)*
1	Beatrice Demonstrator	-	-	-	-	-	-
1	Greater Gabbard	12.4	8	49.6	2.0	62.0	10.0
1	Gunfleet Sands	1.0	0.3	0.0	0.0	1.0	0.3
1	Kentish Flats	-	-	-	-	-	-
1	Kentish Flats Extension	0.3	0.1	1.3	0.1	1.6	0.2
1	Lincs	1.7		6.8	0.3	8.5	0.3
1	London Array	-	-	-	-	-	-
1	Lynn and Inner Dowsing	-	-	-	-	-	-
1	Scroby Sands	-	-	-	-	-	-
1	Sheringham Shoal	1.7	0.3	6.6	0.3	8.3	0.6
1	Teesside	0.0		0.0	0.0	0.0	0.0
1	Thanet	3.2	1.4	12.8	0.5	16.0	1.9
1	Humber Gateway	0.3		1.1	0.0	1.4	0.0
1	Westermost Rough	0.1		0.3	0.0	0.4	0.0
1	Hywind	0.0		0.0	0.0	0.0	0.0
2	Kincardine	0.0		0.0	0.0	0.0	0.0
2	Beatrice	0.0		0.0	0.0	0.0	0.0
2	Dudgeon	7.7	1.1	30.6	1.2	38.3	2.3
2	Galloper	27.8	18.0	111.0	4.4	138.8	22.4
2	Race Bank	43.2		10.8	0.4	54.0	0.4
2	Rampion	1.6		6.3	0.3	7.9	0.3
2	Hornsea Project One	4.4		17.4	0.7	21.8	0.7
3	Blyth Demonstration Project	0.0		0.0	0.0	0.0	0.0
3	Dogger Bank Creyke Beck Projects A and B	2.6		10.4	0.4	13.0	0.4
3	East Anglia ONE	5.9	2.2	33.8	1.4	39.7	3.6
3	European Offshore Wind Deployment Centre	0.0		0.0	0.0	0.0	0.0
3	Firth of Forth Alpha and Bravo	2.1		8.4	0.3	10.5	0.3
3	Inch Cape	0.0		0.0	0.0	0.0	0.0
3	**Methil**	**0.5**		**0.0**	**0.0**	**0.5**	**0.0**

Tier	Wind farm	Breeding season		Nonbreeding season		Annual	
		Total	AOE SPA	Total	AOE SPA	Total	AOE SPA (nonbreeding season apportioned plus breeding season for wind farms <141km)*
3	Moray Firth (EDA)	0.0		0.0	0.0	0.0	0.0
3	Neart na Gaoithe	0.3		1.2	0.0	1.5	0.0
3	Dogger Bank Teesside Projects A and B	2.4		9.6	0.4	12.0	0.4
3	Triton Knoll	7.4		29.6	1.2	37.0	1.2
3	Hornsea Project Two	2.0		2.0	0.1	4.0	0.1
4	East Anglia THREE	1.8	0.4	8.2	0.3	10.0	0.7
5	**Hornsea Project Three**	**17.3**		**0.0**	**0.0**	**17.3**	0.0
5	**Thanet Extension**	**3.0**	**1.3**	**2.0**	**0.1**	**5.0**	**1.4**
5	**Norfolk Vanguard**	**15.6**	**2.7**	**7.5**	**0.3**	**23.1**	**3.0**
6	Moray West	0.0		0.0	0.0	0.0	0.0
6	Norfolk Boreas	17.3	3.6	22.5	0.9	39.8	4.5
6	**East Anglia TWO**	**4.7**	**1.8**	**0.5**	**0.0**	**5.2**	**1.8**
6	**East Anglia ONE North**	**1.0**	**0.2**	**0.6**	**0.0**	**1.6**	**0.2**
6	***Hornsea 4 (PEIR)***	***1.9***		***0.0***	***0.0***	***1.9***	**0.0**
	Total (all projects)	**191.2**	**41.4**	**390.9**	**15.6**	**582.1**	**57.0**
	Total (minus Hornsea Project Three)	**173.9**	**41.4**	**390.9**	**15.6**	**564.8**	**57.0**
	Total (minus Hornsea Project Four)	**189.3**	**41.4**	**390.9**	**15.6**	**580.2**	**57.0**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**172.0**	**41.4**	**390.9**	**15.6**	**562.9**	**57.0**

Table 7.5 Herring gull cumulative collision risk.

Tier	Wind farm	Breeding season	Nonbreeding season	Annual
1	Beatrice Demonstrator	0.0		0.0
1	Greater Gabbard	0.0		0.0
1	Gunfleet Sands	-	-	-
1	**Kentish Flats**	**0.0**	**0**	**0**
1	**Kentish Flats Extension**	**0.5**	**1.7**	**2.2**
1	Lincs	0.0		0.0
1	London Array	-	-	-
1	Lynn and Inner Dowsing	0.0		0.0
1	Scroby Sands	-	-	-
1	Sheringham Shoal	0.0		0.0
1	Teesside	8.7	34.5	43.2
1	Thanet	4.9	19.6	24.5
1	Humber Gateway	0.4	1.1	1.5
1	Westermost Rough	0.1	0.0	0.1
1	Hywind	0.6	7.8	8.4
2	Kincardine	1.0	0.0	1.0
2	Beatrice	49.4	197.4	246.8
2	Dudgeon	-	-	-
2	Galloper	27.2		27.2
2	Race Bank	0.0		0.0
2	Rampion	155.0		155.0
2	Hornsea Project One	2.9	11.6	14.5
3	Blyth Demonstration Project	0.5	2.2	2.7
3	Dogger Bank Creyke Beck Projects A and B	0.0		0.0
3	East Anglia ONE	0.0	28.0	28.0
3	European Offshore Wind Deployment Centre	4.8		4.8
3	Firth of Forth Alpha and Bravo	10.0	21.0	31.0
3	Inch Cape	0.0	13.5	13.5
3	**Methil**	**5.8**	**3.7**	**9.5**
3	Moray Firth (EDA)	52.0		52.0
3	Neart na Gaoithe	5.0	12.5	17.5
3	Dogger Bank Teesside Projects A and B	0.0		0.0
3	Triton Knoll	0.0		0.0
3	Hornsea Project Two	23.8		23.8

Tier	Wind farm	Breeding season	Nonbreeding season	Annual
4	East Anglia THREE	0.0	23.0	23.0
5	Hornsea Project Three	1.0	8.3	9.3
5	**Thanet Extension**	**15.0**	**10.0**	**25.0**
5	**Norfolk Vanguard**	**0.8**	**12.7**	**13.5**
6	Moray West	12.0	1.0	13.0
6	Norfolk Boreas	3.9	14.5	18.4
6	**East Anglia TWO**	**0.0**	**0.5**	**0.5**
6	East Anglia ONE North	0.0	0.0	0.0
6	***Hornsea 4 (PEIR)***	**1.8**	**0.8**	**2.6**
	Total (all projects)	**387.1**	**425.4**	**812.5**
	Total (minus Hornsea Project Three)	**386.1**	**417.1**	**803.2**
	Total (minus Hornsea Project Four)	**385.3**	**424.6**	**809.9**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**384.3**	**416.3**	**800.6**

Table 7.6 Great black-backed gull cumulative collision risk.

Tier	Wind farm	Breeding season	Nonbreeding season	Annual
1	Beatrice Demonstrator	0.0	0.0	0.0
1	Greater Gabbard	15.0	60.0	75.0
1	Gunfleet Sands	-	-	-
1	Kentish Flats	-	-	-
1	Kentish Flats Extension	0.1	0.2	0.3
1	Lincs	0.0	0.0	0.0
1	London Array	-	-	-
1	Lynn and Inner Dowsing	0.0	0.0	0.0
1	Scroby Sands	-	-	-
1	Sheringham Shoal	0.0	0.0	0.0
1	Teesside	8.7	34.8	43.6
1	Thanet	0.1	0.4	0.5
1	Humber Gateway	1.3	5.1	6.3
1	Westermost Rough	0.0	0.0	0.1
1	Hywind	0.3	4.5	4.8
2	Kincardine	0.0	0.0	0.0
2	Beatrice	30.2	120.8	151.0
2	Dudgeon	0.0	0.0	0.0
2	Galloper	4.5	18.0	22.5
2	Race Bank	0.0	0.0	0.0
2	Rampion	5.2	20.8	26.0
2	Hornsea Project One	17.2	68.6	85.8
3	Blyth Demonstration Project	1.3	5.1	6.3
3	Dogger Bank Creyke Beck Projects A and B	5.8	23.3	29.1
3	East Anglia ONE	0.0	46.0	46.0
3	European Offshore Wind Deployment Centre	0.6	2.4	3.0
3	Firth of Forth Alpha and Bravo	13.4	53.4	66.8
3	Inch Cape	0.0	36.8	36.8
3	Methil	0.8	0.8	1.6
3	Moray Firth (EDA)	9.5	25.5	35.0
3	Neart na Gaoithe	0.9	3.6	4.5
3	Dogger Bank Teesside Projects A and B	6.4	25.5	31.9
3	Triton Knoll	24.4	97.6	122.0
3	Hornsea Project Two	3.0	20.0	23.0

Tier	Wind farm	Breeding season	Nonbreeding season	Annual
4	East Anglia THREE	4.6	34.4	39.0
5	Hornsea Project Three	19.4	46.6	66.0
5	**Thanet Extension**	**6.5**	**35.5**	**42.0**
5	**Norfolk Vanguard**	**8.1**	**38.8**	**46.9**
6	Moray West	4.0	5.0	9.0
6	Norfolk Boreas	7.8	85.4	93.1
6	**East Anglia TWO**	**3.8**	**3.7**	**7.5**
6	**East Anglia ONE North**	**3.9**	**1.3**	**5.2**
6	***Hornsea 4 (PEIR)***	***3.0***	***13.6***	***13.6***
	Total (all projects)	**209.8**	**937.5**	**1144.2**
	Total (minus Hornsea Project Three)	**190.4**	**890.9**	**1078.2**
	Total (minus Hornsea Project Four)	**206.8**	**923.9**	**1130.6**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**187.4**	**877.3**	**1064.6**

7.2 Cumulative and In-combination Displacement Risk Tables

Table 7.7 Gannet cumulative and in-combination displacement risk.

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
1	Beatrice Demonstrator	-	-	-	-	-	-	-	-
1	Greater Gabbard	252	0	69	3.3	105	6.5	426	9.8
1	Gunfleet Sands	0	0	12	0.6	9	0.6	21	1.2
1	Kentish Flats	-	-	-	-	-	-	-	-
1	Kentish Flats Extension	0	0	13	0.6	0	0	13	0.6
1	Lincs	-	-	-	-	-	-	-	-
1	London Array	-	-	-	-	-	-	-	-
1	Scroby Sands	-	-	-	-	-	-	-	-
1	Sheringham Shoal	47	47	31	1.5	2	0.1	80	48.6
1	Teesside	1	0.5	0	0	0	0	1	0.5
1	Thanet	-	-	-	-	-	-	-	-
1	Humber Gateway	-	-	-	-	-	-	-	-
1	Westermost Rough	-	-	-	-	-	-	-	-
1	Hywind	10	0	0	0	4	0.2	14	0.2
2	Kincardine	120	0	0	0	0	0	120	0
2	Beatrice	151	0	0	0	0	0	151	0
2	Dudgeon	53	53	25	1.2	11	0.7	89	54.9
2	Galloper	360	0	907	43.5	276	17.1	1543	60.6
2	Race Bank	92	92	32	1.5	29	1.8	153	95.3
2	Rampion	0	0	590	28.3	0	0	590	28.3
2	Hornsea Project One	671	671	694	33.3	250	15.5	1615	719.8
3	Blyth Demonstration Project	-	-	-	-	-	-	-	-
3	Dogger Bank Creyke Beck A	518	259	916	44	176	10.9	1610	313.9
3	Dogger Bank Creyke Beck B	637	318.5	1132	54.3	218	13.5	1987	386.3
3	East Anglia ONE	161	161	3638	174.6	76	4.7	3875	340.3
3	European Offshore Wind Deployment Centre	35	0	5	0.2	0	0	40	0.2
3	Firth of Forth Alpha	1716	0	296	14.2	138	8.6	2150	22.8

Tier	Wind farm	Breeding season		Autumn migration		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
3	Firth of Forth Bravo	1240	0	368	17.7	194	12	1802	29.7
3	Inch Cape	2398	0	703	33.7	212	13.1	3313	46.8
3	**Methil**	**23**	**0**	**0**	**0**	**0**	**0**	**23**	**0**
3	Moray Firth (EDA)	564	0	292	14	27	1.7	883	15.7
3	Neart na Gaoithe	1987	0	552	26.5	281	17.4	2820	43.9
3	Dogger Bank Teesside A	968	484	379	18.2	226	14	1573	516.2
3	Dogger Bank Teesside B	1282	641	508	24.4	238	14.8	2028	680.2
3	Triton Knoll	211	211	15	0.7	24	1.5	250	213.2
3	Hornsea Project Two	457	457	1140	54.7	124	7.7	1721	519.4
4	East Anglia THREE	412	412	1269	60.9	524	32.5	2205	505.4
5	Hornsea Project Three	1203	1203	1494	71.7	1099	68.1	3796	1342.8
5	Thanet Extension	27	0	324	15.6	384	23.8	735	39.4
5	Norfolk Vanguard East	176	176	1630	78.2	419	26	2225	280.2
5	Norfolk Vanguard West	95	95	823	39.5	18	1.1	936	135.6
6	Moray West	2827	0	439	21.1	144	8.9	3410	30
6	Norfolk Boreas	1229	1229	1723	82.7	526	32.6	3478	1344.3
6	**East Anglia TWO**	**192**	**192**	**891**	**42.8**	**192**	**11.9**	**1275**	**246.7**
6	**East Anglia ONE North**	**149**	**149**	**468**	**22.5**	**44**	**2.7**	**661**	**174.2**
6	***Hornsea 4 (PEIR)***	***1892***	***1892***	***1192***	***57.2***	***659***	***40.9***	***3743***	***1990.1***
	Total (all projects)	**22156**	**8743**	**22570**	**1083.2**	**6629**	**410.9**	**51355**	**10237.1**
	Total (minus Hornsea Project Three)20953	**20953**	**7540**	**21076**	**1011.5**	**5530**	**342.8**	**47559**	**8894.3**
	Total (minus Hornsea Project Four)	**20264**	**6851**	**21378**	**1026**	**5970**	**370**	**47612**	**8247**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**19061**	**5648**	**19884**	**954.3**	**4871**	**301.9**	**43816**	**6904.2**

Table 7.8 Guillemot cumulative and in-combination displacement risk.

Tier	Wind farm	Breeding season		Nonbreeding season		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
1	**Beatrice Demonstrator**	**-**	**-**	**-**	**-**	**-**	**-**
1	**Greater Gabbard**	**345**	**0**	**548**	**24.1**	**893**	**24.1**
1	**Gunfleet Sands**	**0**	**0**	**363**	**16**	**363**	**16**
1	**Kentish Flats**	**0**	**0**	**3**	**0.1**	**3**	**0.1**
1	**Kentish Flats Extension**	**0**	**0**	**4**	**0.2**	**4**	
1	Lincs & LID	582	0	814	35.8	1396	35.8
1	London Array	192	0	377	16.6	569	16.6
1	**Scroby Sands**	**-**	**-**	**-**	**-**	**-**	**-**
1	Sheringham Shoal	390	0	715	31.5	1105	31.5
1	Teesside	267	267	901	39.6	1168	306.6
1	Thanet	18	0	124	5.5	142	5.5
1	Humber Gateway	99	99	138	6.1	237	105.1
1	Westermost Rough	347	347	486	21.4	833	368.4
1	Hywind	249	0	2136	94	2385	94
2	Kincardine	632	0	0	0	632	0
2	Beatrice	13610	0	2755	121.2	16365	121.2
2	Dudgeon	334	0	542	23.8	876	23.8
2	Galloper	305	0	593	26.1	898	26.1
2	Race Bank	361	0	708	31.2	1069	31.2
2	**Rampion**	**10887**	**0**	**15536**	**683.6**	**26423**	**683.6**
2	Hornsea Project One	9836	4554.1	8097	356.3	17933	4910.4
3	Blyth Demonstration Project	1220	0	1321	58.1	2541	58.1
3	Dogger Bank Creyke Beck A	5407	1892.5	6142	270.2	11549	2162.7
3	Dogger Bank Creyke Beck B	9479	3317.7	10621	467.3	20100	3785
3	East Anglia ONE	274	0	640	28.2	914	28.2
3	European Offshore Wind Deployment Centre	547	0	225	9.9	772	9.9
3	Firth of Forth Alpha	13606	0	4688	206.3	18294	206.3
3	Firth of Forth Bravo	11118	0	4112	180.9	15230	180.9
3	Inch Cape	4371	0	3177	139.8	7548	139.8
3	**Methil**	**25**	**0**	**0**	**0**	**25**	**0**
3	Moray Firth (EDA)	9820	0	547	24.1	10367	24.1
3	Neart na Gaoithe	1755	0	3761	165.5	5516	165.5
3	Dogger Bank Teesside A	3283	1149.1	2268	99.8	5551	1248.9

Tier	Wind farm	Breeding season		Nonbreeding season		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
3	Dogger Bank Teesside B	5211	1823.9	3701	162.8	8912	1986.7
3	Triton Knoll	425	425	746	32.8	1171	457.8
3	Hornsea Project Two	7735	3581.3	13164	579.2	20899	4160.5
4	East Anglia THREE	1744	0	2859	125.8	4603	125.8
5	Hornsea Project Three	13374	0	19174	843.7	32548	843.7
5	**Thanet Extension**	**12**	**0**	**1105**	**48.6**	**1117**	**48.6**
5	Norfolk Vanguard East	2931	0	2197	96.7	5128	96.7
5	Norfolk Vanguard West	1389	0	2579	113.5	3968	113.5
6	Moray West	24426	0	38174	1679.7	62600	1679.7
6	Norfolk Boreas	7767	0	13777	606.2	21544	606.2
6	**East Anglia TWO**	**2077**	**0**	**1675**	**73.7**	**3752**	**73.7**
6	**East Anglia ONE North**	**4183**	**0**	**1888**	**83.1**	**6071**	**83.1**
6	***Hornsea 4 (PEIR)***	***15245***	*15245*	***69555***	***3060.4***	***84800***	***18305.4***
	Total (all projects)	**185878**	**32701.6**	**242936**	**10689.4**	**428814**	**43390.8**
	Total (minus Hornsea Project Three)	**172504**	**32701.6**	**223762**	**9845.7**	**396266**	**42547.1**
	Total (minus Hornsea Project Four)	**170633**	**17456.6**	**173381**	**7629**	**344014**	**25085.4**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**157259**	**17456.6**	**154207**	**6785.3**	**311466**	**24241.7**

Table 7.9 Razorbill cumulative and in-combination displacement risk.

Tier	Wind farm	Breeding season		Autumn migration		Nonbreeding season		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
1	Beatrice Demonstrator	-	-	-	-	-	-	-	-	-	-
1	Greater Gabbard	0	0	0	0	387	10.5	84	2.8	471	13
1	Gunfleet Sands	0	0	0	0	30	0.8	0	0	30	1
1	Kentish Flats	-	-	-	-	-	-	-	-	-	-
1	Kentish Flats Extension	-	-	-	-	-	-	-	-	-	-
1	Lincs & LID	45	0	34	1.1	22	0.6	34	1.1	134	3
1	London Array	14	0	20	0.7	14	0.4	20	0.7	68	2
1	Scroby Sands	-	-	-	-	-	-	-	-	-	-
1	Sheringham Shoal	106	0	1343	45.7	211	5.7	30	1	1690	52
1	Teesside	16	0	61	2.1	2	0.1	20	0.7	99	3
1	Thanet	3	0	0	0	14	0.4	21	0.7	37	1
1	Humber Gateway	27	0	20	0.7	13	0.4	20	0.7	80	2
1	Westermost Rough	91	91	121	4.1	152	4.1	91	3.1	455	102
1	Hywind	30	0	719	24.4	10	0.3			759	25
2	Kincardine	22	0		0		0			22	0
2	Beatrice	873	0	833	28.3	555	15	833	28.3	3094	72
2	Dudgeon	256	0	346	11.8	745	20.1	346	11.8	1693	44
2	Galloper	44	0	43	1.5	106	2.8	394	13.4	587	18
2	Race Bank	28	0	42	1.4	28	0.8	42	1.4	140	4
2	**Rampion**	**630**	**0**	**66**	**2.2**	**1244**	**33.6**	**3327**	**113.1**	**5267**	**149**

Tier	Wind farm	Breeding season		Autumn migration		Nonbreeding season		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
2	Hornsea Project One	1109	534.5	4812	163.6	1518	41	1803	61.3	9242	800
3	Blyth Demonstration Project	121	0	91	3.1	61	1.6	91	3.1	364	8
3	Dogger Bank Creyke Beck A	1250	375	1576	53.6	1728	46.7	4149	141.1	8703	616
3	Dogger Bank Creyke Beck B	1538	461.4	2097	71.3	2143	57.9	5119	174	10897	765
3	East Anglia ONE	16	0	26	0.9	155	4.2	336	11.4	533	17
3	European Offshore Wind Deployment Centre	161	0	64	2.2	7	0.2	26	0.9	258	3
3	**Firth of Forth Alpha**	**5876**	**0**			**1103**	**29.8**			**6979**	**30**
3	**Firth of Forth Bravo**	**3698**	**0**			**1272**	**34.3**			**4970**	**34**
3	Inch Cape	1436	0	2870	97.6	651	17.6			4957	115
3	**Methil**	**4**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**4**	**0**
3	Moray Firth (EDA)	2423	0	1103	37.5	30	0.8	168	5.7	3724	44
3	Neart na Gaoithe	331	0	5492	186.7	508	13.7			6331	200
3	Dogger Bank Teesside A	834	250.2	310	10.6	959	25.9	1919	65.2	4022	352
3	Dogger Bank Teesside B	1153	345.9	592	20.1	1426	38.5	2953	100.4	6125	505
3	Triton Knoll	40	0	254	8.6	855	23.1	117	4	1265	36
3	Hornsea Project Two	2511	1210.3	4221	143.5	720	19.4	1668	56.7	9119	1430
4	East Anglia THREE	1807	0	1122	38.1	1499	40.5	1524	51.8	5952	130
5	Hornsea Project Three	630	0	2020	68.7	5024	135.6	1754	59.6	9428	264
5	**Thanet Extension**	**0**	**0**	**6**	**0.2**	**56**	**1.5**	**124**	**4.2**	**186**	**6**
5	Norfolk Vanguard East	599	0	491	16.7	491	13.3	752	25.6	2333	56

Tier	Wind farm	Breeding season		Autumn migration		Nonbreeding season		Spring migration		Annual	
		Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
5	Norfolk Vanguard West	280	0	375	12.8	348	9.4	172	5.8	1175	28
6	Moray West	2808	0	3544	120.5	184	5	3585	121.9	10121	247
6	Norfolk Boreas	630	0	263	8.9	1065	28.8	345	11.7	2303	49
6	**East Anglia TWO**	**281**	**0**	**44.1**	**1.5**	**136.4**	**3.7**	**230**	**7.8**	**692**	**13**
6	**East Anglia ONE North**	**403**	**0**	**85**	**2.9**	**54**	**1.5**	**207**	**7**	**749**	**11**
6	***Hornsea 4 (PEIR)***	***580***	***580***	***5960***	***202.6***	***685***	***18.5***	***1361***	***46.3***	***8586***	***847.4***
	Total (all projects)	**32704**	**3848.3**	**41066.1**	**1396.2**	**26211.4**	**708.1**	**33665.0**	**1144.3**	**133644.0**	**7097.4**
	Total (minus Hornsea Project Three)	**32074**	**3848.3**	**39046.1**	**1327.5**	**21187.4**	**572.5**	**31911.0**	**1084.7**	**124216.0**	**6833.4**
	Total (minus Hornsea Project Four)	**32124**	**3268.3**	**35106.1**	**1193.6**	**25526.4**	**689.6**	**32304.0**	**1098.0**	**125058.0**	**6250.0**
	Total (minus Hornsea Project Three and Hornsea Project Four)	**31494**	**3268.3**	**33086.1**	**1124.9**	**20502.4**	**554**	**30550.0**	**1038.4**	**115630.0**	**5985.6**

8 Appendix 3 – NE Seabird PVA Input parameters

321. The Applicant was advised by Natural England during consultations following their review of the original application (links were provided via email on the 10th Sept 2019) to make use of the Seabird PVA tool (https://github.com/naturalengland/Seabird_PVA_Tool). This Appendix provides the input settings used to conduct the population simulations reported in this assessment.

322. It should be noted however, that Natural England subsequently informed the Applicant that there were expected to be changes in the format of the outputs produced by the model and the Applicant was advised that if possible it would be worth waiting for the model to be updated before making use of it. However, owing to the limited amount of time available to the Applicant to undertake the ornithology assessment in this report and the risk of updates to the PVA being delayed, a decision was made to use the model in its current state. It has now been confirmed that the revised version of the model will not be available until January 2020 at the earliest. Furthermore, the Applicant's understanding of the proposed changes is that these relate primarily to how the outputs are presented and not to the underlying structure and mechanics of the model itself. Since this means that the results as presented in this report are unlikely to be materially affected the Applicant considers that the results presented in this report are robust and reliable for the purposes of assessment (the model developer has also informed the Applicant that in their opinion the model structure and outputs are unlikely to change). Nonetheless, the Applicant acknowledges that it will be necessary to confirm this to be the case once the revised model is available, and this will be undertaken as soon as is practical and an update or clarification submitted to the Examination as appropriate.

323. Simulations were undertaken using the web-based version of this tool, with simulations undertaken both with and without density dependence. Density dependent simulations were run with the population regulation effect set to reduce productivity only and a value found as that which generated a stable population size under baseline settings (i.e. with no impacts).

324. In all cases model runs were trialled initially with 100 simulations to ensure the parameters were correctly defined, following which runs with 1,000 and finally 5,000, were attempted. However, for several cases (noted in the individual of files below) the model would not successfully run with the larger numbers of simulations. The reasons for this are not provided by the tool, so it was necessary in those instances to use the outputs obtained from the highest successful model run. While this is not an ideal situation, on the basis of experience from previous population

modelling this is not considered to have had a large effect on the median and mean outputs reported above.

8.1 Gannet – North Sea scale

8.1.1 BDMPS Density Dependent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 21:32:46
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name GX.BDMPS.DD0.013.5000.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: dduloglin.
Include demographic stochasticity in model?: Yes.
Number of simulations: 5000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Northern Gannet.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 1 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 456298 in 2020
Productivity rates: mean: 0.6971315 , sd: 0.08576701 , DD: -0.013
Adult survival rates: mean: 0.919 , sd: 0.042 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.424 , sd: 0.045 , DD: 0
Age class 1 to 2 - mean: 0.829 , sd: 0.026 , DD: 0
Age class 2 to 3 - mean: 0.891 , sd: 0.019 , DD: 0
Age class 3 to 4 - mean: 0.895 , sd: 0.019 , DD: 0
Age class 4 to 5 - mean: 0.919 , sd: 0.042 , DD: 0
Impacts
Number of impact scenarios: 5.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051

Impact on Demographic Rates
Scenario A - Name: run2800
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006136341 , se: NA
Scenario B - Name: run2900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006355496 , se: NA
Scenario C - Name: run3000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006574651 , se: NA
Scenario D - Name: run3100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006793806 , se: NA
Scenario E - Name: run3200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.007012961 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.1.2 Biogeographic Density Dependent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 21:43:17
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
```
##             Package          Version
## popbio       "popbio"        "2.4.4"
## shiny        "shiny"         "1.1.0"
## shinyjs      "shinyjs"       "1.0"
## shinydashboard "shinydashboard" "0.7.1"
## shinyWidgets  "shinyWidgets"  "0.4.5"
## DT           "DT"            "0.5"
## plotly       "plotly"        "4.8.0"
## rmarkdown    "rmarkdown"     "1.10"
## dplyr        "dplyr"         "0.7.6"
## tidyr        "tidyr"         "0.8.1"
```
Basic information
This run had reference name GX.biogeo.DD0.013.5000.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: dduloglin.
Include demographic stochasticity in model?: Yes.
Number of simulations: 5000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates

Species chosen to set initial values: Northern Gannet.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 1 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 1180000 in 2020
Productivity rates: mean: 0.6971315 , sd: 0.08576701 , DD: -0.013
Adult survival rates: mean: 0.919 , sd: 0.042 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.424 , sd: 0.045 , DD: 0
Age class 1 to 2 - mean: 0.829 , sd: 0.026 , DD: 0
Age class 2 to 3 - mean: 0.891 , sd: 0.019 , DD: 0
Age class 3 to 4 - mean: 0.895 , sd: 0.019 , DD: 0
Age class 4 to 5 - mean: 0.919 , sd: 0.042 , DD: 0
Impacts
Number of impact scenarios: 5.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: run2800
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002373 , se: NA
Scenario B - Name: run2900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002458 , se: NA
Scenario C - Name: run3000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002542 , se: NA
Scenario D - Name: run3100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002627 , se: NA
Scenario E - Name: run3200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002712 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.1.3 BDMPS Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 21:56:26
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)

```
##               Package         Version
## popbio        "popbio"        "2.4.4"
## shiny         "shiny"         "1.1.0"
## shinyjs       "shinyjs"       "1.0"
## shinydashboard "shinydashboard" "0.7.1"
## shinyWidgets  "shinyWidgets"  "0.4.5"
## DT            "DT"            "0.5"
## plotly        "plotly"        "4.8.0"
## rmarkdown     "rmarkdown"     "1.10"
## dplyr         "dplyr"         "0.7.6"
## tidyr         "tidyr"         "0.8.1"
```

Basic information
This run had reference name GX.BDMPS.DI.5000.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 5000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Northern Gannet.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 1 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 456298 in 2020
Productivity rates: mean: 0.6971315 , sd: 0.08576701 , DD: 0
Adult survival rates: mean: 0.919 , sd: 0.042 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.424 , sd: 0.045 , DD: 0
Age class 1 to 2 - mean: 0.829 , sd: 0.026 , DD: 0
Age class 2 to 3 - mean: 0.891 , sd: 0.019 , DD: 0
Age class 3 to 4 - mean: 0.895 , sd: 0.019 , DD: 0
Age class 4 to 5 - mean: 0.919 , sd: 0.042 , DD: 0
Impacts
Number of impact scenarios: 5.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: run2800

All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006136341 , se: NA
Scenario B - Name: run2900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006355496 , se: NA
Scenario C - Name: run3000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006574651 , se: NA
Scenario D - Name: run3100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.006793806 , se: NA
Scenario E - Name: run3200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.007012961 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.1.4 Biogeographic Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 21:17:02
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name GX.biogeo.DI.5000.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 5000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Northern Gannet.
Region type to use for breeding success data: Global.

Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 1 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 1180000 in 2020
Productivity rates: mean: 0.6971315 , sd: 0.08576701 , DD: 0
Adult survival rates: mean: 0.919 , sd: 0.042 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.424 , sd: 0.045 , DD: 0
Age class 1 to 2 - mean: 0.829 , sd: 0.026 , DD: 0
Age class 2 to 3 - mean: 0.891 , sd: 0.019 , DD: 0
Age class 3 to 4 - mean: 0.895 , sd: 0.019 , DD: 0
Age class 4 to 5 - mean: 0.919 , sd: 0.042 , DD: 0
Impacts
Number of impact scenarios: 5.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: run2800
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002373 , se: NA
Scenario B - Name: run2900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002458 , se: NA
Scenario C - Name: run3000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002542 , se: NA
Scenario D - Name: run3100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002627 , se: NA
Scenario E - Name: run3200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002712 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.2 Kittiwake – North Sea scale

8.2.1 BDMPS Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-12-02 17:56:01
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name Ki BDMPS 500sims 3900to4400.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 500.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Black-Legged Kittiwake.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 4.
Is there an upper constraint on productivity in the model?: Yes, constrained to 2 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 829937 in 2020
Productivity rates: mean: 0.6036278 , sd: 0.325783
Adult survival rates: mean: 0.854 , sd: 0.077
Immatures survival rates:
Age class 0 to 1 - mean: 0.79 , sd: 0.077 , DD: NA
Age class 1 to 2 - mean: 0.854 , sd: 0.077 , DD: NA
Age class 2 to 3 - mean: 0.854 , sd: 0.077 , DD: NA
Age class 3 to 4 - mean: 0.854 , sd: 0.077 , DD: NA
Impacts
Number of impact scenarios: 6.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates

Scenario A - Name: 3900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.004699152 , se: NA
Scenario B - Name: 4000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.004819643 , se: NA
Scenario C - Name: 4100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.004940134 , se: NA
Scenario D - Name: 4200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.005060625 , se: NA
Scenario E - Name: 4300
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.005181116 , se: NA
Scenario F - Name: 4400
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.005301607 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.adults
Target population size to use in calculating impact metrics: NA

8.2.2 Quasi-extinction threshold to use in calculating impact metrics: NA Biogeographic Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-12-02 18:01:44
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)

```
##              Package         Version
## popbio       "popbio"        "2.4.4"
## shiny        "shiny"         "1.1.0"
## shinyjs      "shinyjs"       "1.0"
## shinydashboard "shinydashboard" "0.7.1"
## shinyWidgets "shinyWidgets"  "0.4.5"
## DT           "DT"            "0.5"
## plotly       "plotly"        "4.8.0"
## rmarkdown    "rmarkdown"     "1.10"
## dplyr        "dplyr"         "0.7.6"
## tidyr        "tidyr"         "0.8.1"
```

Basic information
This run had reference name Ki biogeo 500sims 3900to4400.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 500.
Random seed: 1.

Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Black-Legged Kittiwake.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 4.
Is there an upper constraint on productivity in the model?: Yes, constrained to 2 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 5100000 in 2020
Productivity rates: mean: 0.6036278 , sd: 0.325783
Adult survival rates: mean: 0.854 , sd: 0.077
Immatures survival rates:
Age class 0 to 1 - mean: 0.79 , sd: 0.077 , DD: NA
Age class 1 to 2 - mean: 0.854 , sd: 0.077 , DD: NA
Age class 2 to 3 - mean: 0.854 , sd: 0.077 , DD: NA
Age class 3 to 4 - mean: 0.854 , sd: 0.077 , DD: NA
Impacts
Number of impact scenarios: 6.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: 3900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.000764706 , se: NA
Scenario B - Name: 4000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.000784314 , se: NA
Scenario C - Name: 4100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.000803922 , se: NA
Scenario D - Name: 4200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.000823529 , se: NA
Scenario E - Name: 4300
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.000843137 , se: NA
Scenario F - Name: 4400
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.000862745 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.adults

Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.3 Lesser black-backed gull

8.3.1 BDMPS Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-12-02 20:29:39
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name LB BDMPS 1000sims 500to600.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 1000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Lesser Black-Backed Gull.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 3 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 209007 in 2020
Productivity rates: mean: 0.4000474 , sd: 0.3759093
Adult survival rates: mean: 0.885 , sd: 0.056
Immatures survival rates:
Age class 0 to 1 - mean: 0.82 , sd: 0.056 , DD: NA
Age class 1 to 2 - mean: 0.885 , sd: 0.056 , DD: NA
Age class 2 to 3 - mean: 0.885 , sd: 0.056 , DD: NA
Age class 3 to 4 - mean: 0.885 , sd: 0.056 , DD: NA
Age class 4 to 5 - mean: 0.885 , sd: 0.056 , DD: NA
Impacts
Number of impact scenarios: 2.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051

Impact on Demographic Rates
Scenario A - Name: 500
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002392264 , se: NA
Scenario B - Name: 600
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002870717 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.adults
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.4 Great black-backed gull – North Sea scale

8.4.1 BDMPS Density Dependent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 22:01:30
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name GB DD test.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: dduloglin.
Include demographic stochasticity in model?: Yes.
Number of simulations: 1000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Great Black-Backed Gull.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 3 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 91399 in 2020
Productivity rates: mean: 0.9707373 , sd: 0.435337 , DD: -0.053
Adult survival rates: mean: 0.93 , sd: 0.025 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 1 to 2 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 2 to 3 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 3 to 4 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 4 to 5 - mean: 0.93 , sd: 0.025 , DD: 0
Impacts
Number of impact scenarios: 3.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051

Impact on Demographic Rates
Scenario A - Name: 1000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01094104 , se: NA
Scenario B - Name: 1100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01203514 , se: NA
Scenario C - Name: 1200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01312925 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.4.2 Biogeographic Density Dependent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 22:04:53
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
```
## Package Version
## popbio "popbio" "2.4.4"
## shiny "shiny" "1.1.0"
## shinyjs "shinyjs" "1.0"
## shinydashboard "shinydashboard" "0.7.1"
## shinyWidgets "shinyWidgets" "0.4.5"
## DT "DT" "0.5"
## plotly "plotly" "4.8.0"
## rmarkdown "rmarkdown" "1.10"
## dplyr "dplyr" "0.7.6"
## tidyr "tidyr" "0.8.1"
```
Basic information
This run had reference name GB biogeo DD0.053 1000sims 1000to1200.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: dduloglin.
Include demographic stochasticity in model?: Yes.
Number of simulations: 1000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Great Black-Backed Gull.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 3 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1

Initial population values: Initial population 235000 in 2020
Productivity rates: mean: 0.9707373 , sd: 0.435337 , DD: -0.053
Adult survival rates: mean: 0.93 , sd: 0.025 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 1 to 2 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 2 to 3 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 3 to 4 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 4 to 5 - mean: 0.93 , sd: 0.025 , DD: 0
Impacts
Number of impact scenarios: 3.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: 1000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.004255319 , se: NA
Scenario B - Name: 1100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.004680851 , se: NA
Scenario C - Name: 1200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.005106383 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.4.3 BDMPS Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-10-17 21:49:24
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
```
##              Package          Version
## popbio        "popbio"         "2.4.4"
## shiny         "shiny"          "1.1.0"
## shinyjs       "shinyjs"        "1.0"
## shinydashboard "shinydashboard" "0.7.1"
## shinyWidgets  "shinyWidgets"   "0.4.5"
## DT            "DT"             "0.5"
## plotly        "plotly"         "4.8.0"
## rmarkdown     "rmarkdown"      "1.10"
## dplyr         "dplyr"          "0.7.6"
## tidyr         "tidyr"          "0.8.1"
```
Basic information
This run had reference name GB BDMPS 1000sims 1000to 1200.
PVA model run type: simplescenarios.

Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 1000.
Random seed: 10.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Great Black-Backed Gull.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 3 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 91399 in 2020
Productivity rates: mean: 0.9707373 , sd: 0.435337
Adult survival rates: mean: 0.93 , sd: 0.025
Immatures survival rates:
Age class 0 to 1 - mean: 0.93 , sd: 0.025 , DD: NA
Age class 1 to 2 - mean: 0.93 , sd: 0.022 , DD: NA
Age class 2 to 3 - mean: 0.93 , sd: 0.025 , DD: NA
Age class 3 to 4 - mean: 0.93 , sd: 0.025 , DD: NA
Age class 4 to 5 - mean: 0.93 , sd: 0.025 , DD: NA
Impacts
Number of impact scenarios: 3.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: 1000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01094104 , se: NA
Scenario B - Name: 1100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01203514 , se: NA
Scenario C - Name: 1200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01312925 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.4.4 Biogeographic Density Dependent

NE PVA Parameter log

Set up
The log file was created on: 2019-10-17 22:01:30
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)

```
##              Package         Version
## popbio       "popbio"        "2.4.4"
## shiny        "shiny"         "1.1.0"
## shinyjs      "shinyjs"       "1.0"
## shinydashboard "shinydashboard" "0.7.1"
## shinyWidgets "shinyWidgets"  "0.4.5"
## DT           "DT"            "0.5"
## plotly       "plotly"        "4.8.0"
## rmarkdown    "rmarkdown"     "1.10"
## dplyr        "dplyr"         "0.7.6"
## tidyr        "tidyr"         "0.8.1"
```

Basic information
This run had reference name GB DD test.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: dduloglin.
Include demographic stochasticity in model?: Yes.
Number of simulations: 1000.
Random seed: 1.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Great Black-Backed Gull.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 5.
Is there an upper constraint on productivity in the model?: Yes, constrained to 3 per pair.
Number of subpopulations: 1.
Format for initial population size: all.individuals
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 91399 in 2020
Productivity rates: mean: 0.9707373 , sd: 0.435337 , DD: -0.053
Adult survival rates: mean: 0.93 , sd: 0.025 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 1 to 2 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 2 to 3 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 3 to 4 - mean: 0.93 , sd: 0.025 , DD: 0
Age class 4 to 5 - mean: 0.93 , sd: 0.025 , DD: 0
Impacts
Number of impact scenarios: 3.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: 1000
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01094104 , se: NA
Scenario B - Name: 1100

All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01203514 , se: NA
Scenario C - Name: 1200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.01312925 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.pairs
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

8.5 Guillemot – FFC SPA

8.5.1 Density Independent

NE PVA Parameter log
Set up
The log file was created on: 2019-12-03 14:56:07
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name GU FFC 100 to 3050.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: nodd.
Include demographic stochasticity in model?: Yes.
Number of simulations: 500.
Random seed: 10.
Case study selected: None.
Baseline demographic rates
Species chosen to set initial values: Common Guillemot.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 6.
Is there an upper constraint on productivity in the model?: Yes, constrained to 1 per pair.
Number of subpopulations: 1.
Format for initial population size: breeding.adults
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 83214 in 2020
Productivity rates: mean: 0.5826832 , sd: 0.1894517
Adult survival rates: mean: 0.94 , sd: 0.025
Immatures survival rates:
Age class 0 to 1 - mean: 0.56 , sd: 0.058 , DD: NA
Age class 1 to 2 - mean: 0.792 , sd: 0.152 , DD: NA
Age class 2 to 3 - mean: 0.917 , sd: 0.098 , DD: NA
Age class 3 to 4 - mean: 0.938 , sd: 0.107 , DD: NA
Age class 4 to 5 - mean: 0.94 , sd: 0.025 , DD: NA
Age class 5 to 6 - mean: 0.94 , sd: 0.025 , DD: NA
Impacts
Number of impact scenarios: 5.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative

Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: 100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.001201721 , se: NA
Scenario B - Name: 200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002403442 , se: NA
Scenario C - Name: 1700
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.02042926 , se: NA
Scenario D - Name: 2900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.03484991 , se: NA
Scenario E - Name: 3050
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.03665249 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.adults
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

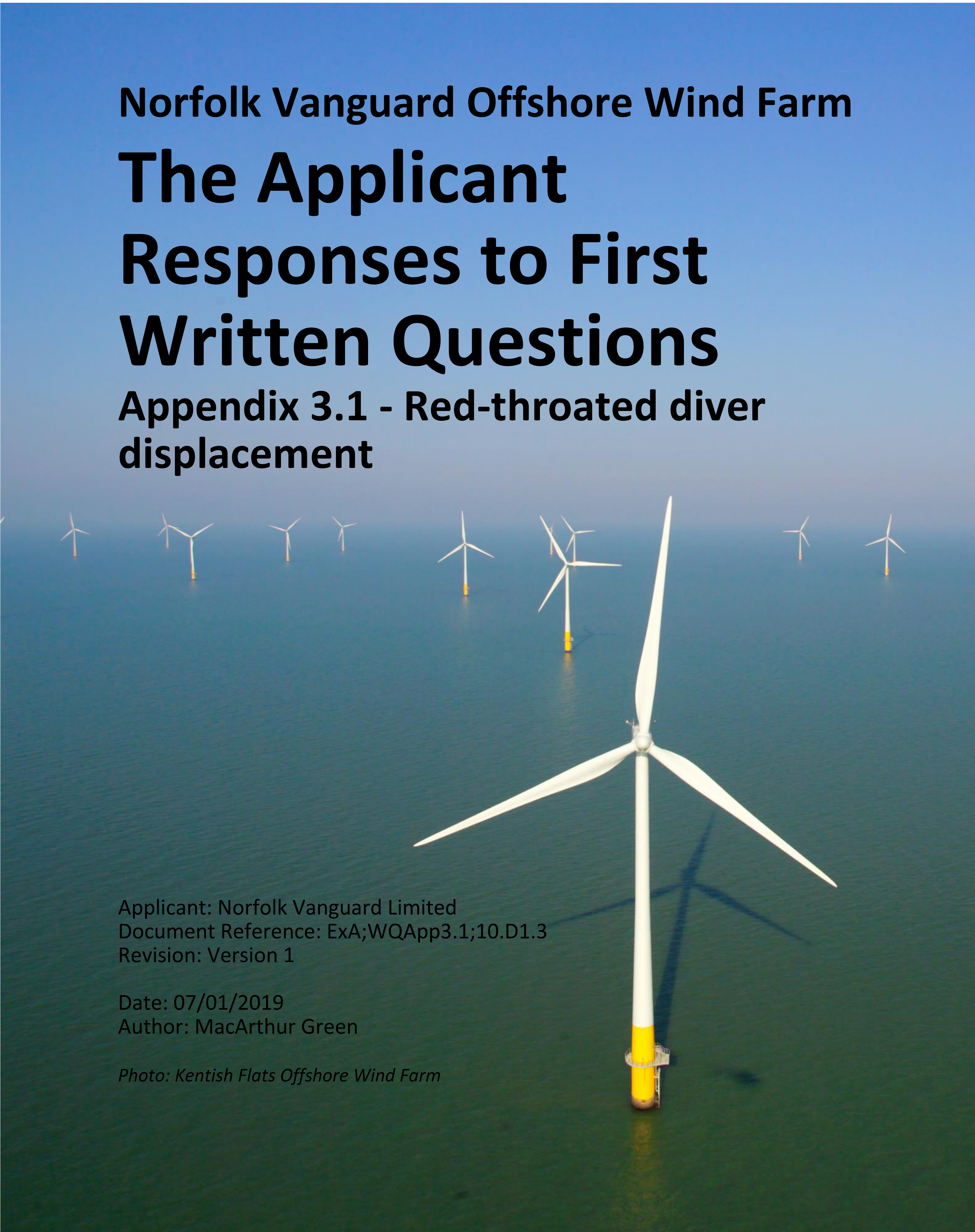
8.5.2 Density Dependent

NE PVA Parameter log
Set up
The log file was created on: 2019-12-03 15:06:08
R version 3.5.1, NEPVA package version: 3.6 (with UI version 1.4)
Package Version
popbio "popbio" "2.4.4"
shiny "shiny" "1.1.0"
shinyjs "shinyjs" "1.0"
shinydashboard "shinydashboard" "0.7.1"
shinyWidgets "shinyWidgets" "0.4.5"
DT "DT" "0.5"
plotly "plotly" "4.8.0"
rmarkdown "rmarkdown" "1.10"
dplyr "dplyr" "0.7.6"
tidyr "tidyr" "0.8.1"
Basic information
This run had reference name GU FFC DD100to3050.
PVA model run type: simplescenarios.
Model to use for environmental stochasticity: betagamma.
Model for density dependence: dduloglin.
Include demographic stochasticity in model?: Yes.
Number of simulations: 500.
Random seed: 15.
Case study selected: None.

Baseline demographic rates
Species chosen to set initial values: Common Guillemot.
Region type to use for breeding success data: Global.
Sector to use within breeding success region: Global.
Age at first breeding: 6.
Is there an upper constraint on productivity in the model?: Yes, constrained to 1 per pair.
Number of subpopulations: 1.
Format for initial population size: breeding.adults
Are demographic rates applied separately to each subpopulation?: Yes.
Are baseline demographic rates specified separately for immatures?: Yes.
Population 1
Initial population values: Initial population 83214 in 2020
Productivity rates: mean: 0.5826832 , sd: 0.1894517 , DD: -0.045
Adult survival rates: mean: 0.94 , sd: 0.025 , DD: 0
Immatures survival rates:
Age class 0 to 1 - mean: 0.56 , sd: 0.058 , DD: 0
Age class 1 to 2 - mean: 0.792 , sd: 0.152 , DD: 0
Age class 2 to 3 - mean: 0.917 , sd: 0.098 , DD: 0
Age class 3 to 4 - mean: 0.938 , sd: 0.107 , DD: 0
Age class 4 to 5 - mean: 0.94 , sd: 0.025 , DD: 0
Age class 5 to 6 - mean: 0.94 , sd: 0.025 , DD: 0
Impacts
Number of impact scenarios: 5.
Are impacts applied separately to each subpopulation?: No
Are impacts of scenarios specified separately for immatures?: No
Are standard errors of impacts available?: No
Should random seeds be matched for impact scenarios?: No
Are impacts specified as a relative value or absolute harvest?: relative
Years in which impacts are assumed to begin and end: 2021 to 2051
Impact on Demographic Rates
Scenario A - Name: 100
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.001201721 , se: NA
Scenario B - Name: 200
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.002403442 , se: NA
Scenario C - Name: 1700
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.02042926 , se: NA
Scenario D - Name: 2900
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.03484991 , se: NA
Scenario E - Name: 3050
All subpopulations
Impact on productivity rate mean: 0 , se: NA
Impact on adult survival rate mean: 0.03665249 , se: NA
Output:
First year to include in outputs: 2021
Final year to include in outputs: 2051
How should outputs be produced, in terms of ages?: breeding.adults
Target population size to use in calculating impact metrics: NA
Quasi-extinction threshold to use in calculating impact metrics: NA

Norfolk Vanguard REP1-008: Appendix 3.1 Red-throated diver displacement

Norfolk Vanguard Limited Reference: ExA; WQApp 3.1;10.D1.3: Cited in this document as MacArthur Green 2019d.



Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.1 - Red-throated diver displacement

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.1;10.D1.3
Revision: Version 1

Date: 07/01/2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
23/11/2018	01D	First draft for Norfolk Vanguard Ltd review	MT	JKL	EV
28/11/2018	02D	Second draft for Norfolk Vanguard Ltd review	MT	RWF	EV
18/12/2018	03D	Third draft for Norfolk Vanguard Ltd review	MT	RWF	EV

EXECUTIVE SUMMARY

This report provides an updated assessment of potential displacement impacts on red-throated divers in relation to the Norfolk Vanguard Offshore Wind Farm. This has been produced taking into account comments provided by Natural England in their Relevant Representation for the project.

Natural England recommended use of a displacement rate of 100% and consequent mortality of 10% including all birds within 4 km of the wind farm boundary, as per Statutory Nature Conservation Body (SNCB) guidelines. A review of studies conducted at offshore wind farms and other relevant literature has been conducted to further inform this assessment. Following this review, Norfolk Vanguard Ltd proposes an evidence-based displacement rate of 90% and a consequent mortality rate of 1%, including birds within 2km of the wind farm boundary. The assessment presented in this report provides both the precautionary rates recommended by Natural England and the evidence-based ones.

This report provides an update of the conclusions of the original assessment following the revised displacement predictions. Most impacts remain of minor significance, as concluded in the Environmental Statement (ES), however minor to moderate significant effects are now predicted for Norfolk Vanguard West in mid-winter and annually, for both NV East and West combined and cumulative. However, it should be noted that these results only apply when the highly precautionary approach recommended by the Statutory Nature Conservation Bodies (SNCBs) is used.

Taking all of the evidence into account (as presented in the review appended to this report), displacement of red-throated divers, from Norfolk Vanguard alone and cumulatively, assessed for the complete south west North Sea red-throated diver Biologically Defined Minimum Population Scale (BDMPS), are still considered to be of minor significance, as concluded in the ES.

Table of Contents

Tables

Glossary

AIS	Automatic Identification System
BACI	Before-After-Control-Impact
BDMPS	Biologically Defined Minimum Population Scale
EIA	Environmental Impact Assessment
ES	Environmental Statement
HRA	Habitats Regulations Assessment
NE	Natural England
NV	Norfolk Vanguard
RTD	Red-throated diver
SNCB	Statutory Nature Conservation Body
SPA / pSPA	Special Protection Area / proposed Special Protection Area

1 INTRODUCTION

1. This report provides an updated assessment of potential red-throated displacement (RTD) from Norfolk Vanguard (NV) alone and cumulatively during both the construction and operational phases of the project with respect to the Environmental Impact Assessment (EIA). Assessment with regards the designated populations of the Greater Wash Special Protection Area (SPA) and Outer Thames proposed SPA (pSPA) are provided in a separate update to the Habitats Regulations Assessment (HRA). This assessment has been produced to address comments provided by Natural England in their Relevant Representation (Natural England 2018), which have been reproduced in Table 1.1.

Table 1.1 Natural England (2018) comments with respect to the assessment of potential impact of displacement on red-throated diver and sections in this report where they have been addressed.

NE Relevant Representation paragraph no.	Comment	Section where addressed
4.2.5	As advised in our Section 42 (Preliminary Environmental Information Report) response, NE require that the variability (uncertainty) in the underlying population estimates (i.e. through consideration of appropriately calculated upper and lower confidence intervals) is considered in the displacement assessments. Whilst the upper and lower confidence limits around the bird abundance estimates are presented in the tables in Annex 1 of Appendix 13.01, these have not been considered in the impact assessments for construction or operational displacement within the ES, with only the mean peak seasonal abundances considered. This approach needs to be revisited for all relevant species.	This has been provided in Table 1.8. Note that in the interests of clarity, the updated displacement matrices only present the mean population estimates, but Table 1.8 and the discussion provides additional consideration of the impacts predicted for the upper and lower confidence interval estimates.

NE Relevant Representation paragraph no.	Comment	Section where addressed
4.2.6	Natural England has a number of concerns with the assessment of displacement impacts on red-throated diver (RTD), including the following key points: a) The mean peak seasonal abundances used by the Applicant in the operational displacement assessments and matrices for Vanguard West appear to be too low, seemingly because only data for birds on the water have been used. The joint Statutory Nature Conservation Bodies (SNCB) interim displacement advice note (SNCBs 2017) advises that displacement assessments should use bird data for birds sitting on the water and birds in flight, as is the case with the Vanguard East assessment. b) In particular, Natural England does not consider the 80% displacement and 5% mortality rate used by the Applicant to be appropriate for assessing disturbance and displacement impacts to RTD from offshore wind farms. We note that this does not follow SNCB guidance (SNCBs 2017). Natural England considers that there is no clear justification to change our current advice of a 4 km buffer and 100% displacement across this buffer, and continue to advise that assessments of operational disturbance and displacement for RTD for offshore wind farm assessments are based on a constant displacement rate across the offshore wind farm site and a 4 km buffer and suggest that a range of displacement rates up to 100% and a mortality rate of up to 10% are considered. These values should also be used in the assessment of construction disturbance and displacement for RTD for both EIA and for the HRA assessment for RTD at the Greater Wash SPA. c) Impacts from the operational phase of the development through vessel movements etc., and how these impacts might be mitigated have not been given sufficient consideration. This applies to the Greater Wash SPA and potentially also the Outer Thames Estuary SPA red-throated diver population, as this also could be affected by vessels transiting the site (as the operations and maintenance port is yet to be confirmed).	a) abundance estimates for NV West have been updated to include birds on the water and in flight throughout. b) The SNCB recommended rates of 100% displacement and 10% mortality within the 4 km buffer have been discussed throughout and used for assessing construction and operational effects. We have also conducted a thorough evidence review which is included in Annex 1. This review concludes that 90% displacement and 1% consequent mortality is a more appropriately precautionary combination for this species and that this effect extends to 1.5 km from the wind farm boundary. The impact using these values is also discussed. Note that effects in relation to the SPA populations (i.e. HRA) will be considered in a separate document. c) Extra consideration for this impact has been provided in Section 1.1.2.

NE Relevant Representation paragraph no.	Comment	Section where addressed
4.2.15	The cumulative RTD displacement mortality has been conducted by the Applicant using the same magnitudes of displacement (80%) and mortality (5%) applied to all birds within the 4 km wind farm buffer. As with the assessment of operational displacement for Vanguard alone, Natural England does not consider this to be precautionary and advises that a worst case scenario of 100% displacement and 10% mortality is used.	The SNCB recommended rates of 100% displacement and 10% mortality including the 4 km buffer have been discussed throughout. We have also conducted a thorough evidence review which is included as an annex. This review concludes that 90% displacement and 1% consequent mortality is a more appropriately precautionary combination for this species, extending within a 1.5 km buffer (although a 2km buffer has been used in this assessment). The impact using these values is also discussed.
4.2.16	The Applicant has considered that all wind farms at which turbines were installed before or during 2012 form part of the Norfolk Vanguard baseline. Natural England does not agree that these wind farms should be considered part of the baseline. This is because, although some of the wind farms included in the Applicant's list have been operational for over 10 years, the RTD population data used in Furness (2015) pre-date the installations. We suggest that a similar approach to that undertaken for the auk cumulative displacement assessments is undertaken for RTD, i.e. to sum the bird abundance estimates for each relevant offshore wind farm and put this total through a displacement matrix, and then assess with a worst case scenario of 100% displacement and 10% mortality. The assessment should include all offshore wind farms located within the south-west North Sea RTD BDMPS.	The red-throated diver assessment has been recalculated including all wind farms in the south west North Sea and using rates recommended by the SNCBs and also those identified by the evidence review (Annex 1).

1.1 Assessment of potential impacts

1.1.1 Potential impacts during operation

1.1.1.1 Project alone: Disturbance and displacement from offshore infrastructure

2. The Norfolk Vanguard ES (Vattenfall 2018) assessed red-throated diver displacement effects using a wind farm displacement rate of 80% and a consequent increase in mortality of 5%, applied to all birds within 4 km of the wind farm boundary.

3. Natural England (2018) states that they do not agree these are appropriate rates and note that these rates are not in line with those recommended which are 100% displacement (up to 4 km from the wind farm) and mortality of 10% (SNCBs 2017). Natural England also identified an error in the populations assessed for NV West, which omitted birds in flight.

4. Following receipt of Natural England (2018), a comprehensive review of the most up to date evidence on red-throated diver disturbance was conducted (Annex 1). This review reaches the following evidence-based conclusions on appropriate rates for displacement and consequent mortality:

- Displacement from offshore wind farms is expected to be 90%;
- The extent of displacement beyond the wind farm boundary is variable, but a figure of 90% within 1.5 km is suitable precautionary; and
- The consequence of displacement is currently unknown (in terms of potential increases in mortality), however considering the ecology of red-throated divers and evidence for displacement effects across a wide range of species, it is likely that any increase in mortality will be close to 0% and is highly unlikely to exceed a precautionary increase of 1%.

5. Combining all of the above, the following sections provide the following revised assessment for red-throated divers:

- Operational displacement matrices for NV East and NV West which highlight both NE's recommended rates (100% & 10% including the 4 km buffer) and those identified in the evidence review (90% and 1% including the 4km buffer) and discussion are provided for effects within the 2 km buffer, in line with the conclusions of the evidence review. Note that the review provides evidence for effects only within 1.5 km, but effects have been assessed using the existing 2 km densities, thereby including precaution on this aspect.
- For both NV East (section 1.1.1.1.1) and NV West (section 1.1.1.1.2) the populations at risk include birds recorded on the water and in flight (correcting the omission of birds in flight for NV West).
- An additional table (Table 1.8) presenting displacement mortality using the upper and lower confidence intervals of the seasonal abundance has been included.

6. The displacement matrices have been populated with data for red-throated diver during the autumn migration, mid-winter and spring migration periods within the site and 4 km buffer. These tables present displacement from 0 – 100% at 10%

increments and mortality from 0 – 100%, at 1% increments up to 10%, and larger gaps thereafter. Shading has been used to highlight the 90-100% displacement and 1-10% mortality ranges defined above. Additional discussion of the potential impact for the evidence-based effect operating within 2 km has also been provided.

1.1.1.1.1 Norfolk Vanguard East

Autumn migration

7. Using the seasonal mean peak autumn migration abundance on NV East and 4 km buffer of 50, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

   - Five individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.2);
   - No individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.2).

8. The BDMPS for red-throated diver in autumn is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of five individuals to this would increase the mortality rate by 0.16%, while the evidence-based prediction (0) results in no increase in mortality.

9. It should also be noted that the mean abundance within NV East and the 2 km buffer was 45 individuals (compared with 50 within the 4 km buffer) which would slightly reduce the magnitude of impact (i.e. by 10%).

10. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.2 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard East (and 4 km buffer) during the autumn migration season that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	0	0	0	0	0	1
2	0	0	0	0	1	1	1	1	1	1
3	0	0	0	1	1	1	1	1	1	2
4	0	0	1	1	1	1	1	2	2	2
5	0	1	1	1	1	2	2	2	2	3
6	0	1	1	1	2	2	2	2	3	3
7	0	1	1	1	2	2	2	3	3	4
8	0	1	1	2	2	2	3	3	4	4
9	0	1	1	2	2	3	3	4	4	5
10	1	1	2	2	3	3	4	4	5	5
20	1	2	3	4	5	6	7	8	9	10
30	2	3	5	6	8	9	11	12	14	15
50	3	5	8	10	13	15	18	20	23	25
75	4	8	11	15	19	23	26	30	34	38
100	5	10	15	20	25	30	35	40	45	50

Mid-winter

11. Using the seasonal mean peak winter abundance on NV East and 4 km buffer of 25, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- Three individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.3);
- No individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.3).

12. The BDMPS for red-throated diver in winter is 10,177 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 2,320 (10,177 x 0.228). The addition of three individuals to this would increase the mortality rate by 0.13%, while the evidence-based prediction (0) results in no increase in mortality.

13. It should be noted that the mean abundance within NV East and the 2 km buffer was 13 individuals (compared with 25 within the 4 km buffer) which would halve the magnitude of predicted impact to 2 individuals using the SNCB recommended rates.

14. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.3 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard East (and 4km buffer) during the winter that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	1
3	0	0	0	0	0	0	1	1	1	1
4	0	0	0	0	1	1	1	1	1	1
5	0	0	0	1	1	1	1	1	1	1
6	0	0	0	1	1	1	1	1	1	2
7	0	0	1	1	1	1	1	1	2	2
8	0	0	1	1	1	1	1	2	2	2
9	0	0	1	1	1	1	2	2	2	2
10	0	1	1	1	1	2	2	2	2	3
20	1	1	2	2	3	3	4	4	5	5
30	1	2	2	3	4	5	5	6	7	8
50	1	3	4	5	6	8	9	10	11	13
75	2	4	6	8	9	11	13	15	17	19
100	3	5	8	10	13	15	18	20	23	25

Spring migration

15. Using the seasonal mean peak autumn migration abundance on NV East and 4 km buffer of 119, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- 12 individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.4);
- One individual at the evidence-based rates of 90% displacement and 1% mortality (Table 1.4).

16. The BDMPS for red-throated diver in spring is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of 12

individuals to this would increase the mortality rate by 0.4%, while the evidence-based prediction (1) results in an increase in mortality of 0.03%.

17. It should be noted that the mean abundance within NV East and the 2 km buffer was 90 individuals (compared with 119 within the 4 km buffer) which would reduce the magnitude of predicted impact by 25%, to 8 individuals using the SNCB recommended rates.

18. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.4 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard East (and 4 km buffer) during the spring migration period that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	1	1	1	1	1	1
2	0	0	1	1	1	1	2	2	2	2
3	0	1	1	1	2	2	2	3	3	4
4	0	1	1	2	2	3	3	4	4	5
5	1	1	2	2	3	4	4	5	5	6
6	1	1	2	3	4	4	5	6	6	7
7	1	2	2	3	4	5	6	7	7	8
8	1	2	3	4	5	6	7	8	9	10
9	1	2	3	4	5	6	7	9	10	11
10	1	2	4	5	6	7	8	10	11	12
20	2	5	7	10	12	14	17	19	21	24
30	4	7	11	14	18	21	25	29	32	36
50	6	12	18	24	30	36	42	48	54	60
75	9	18	27	36	45	54	62	71	80	89
100	12	24	36	48	60	71	83	95	107	119

Complete nonbreeding season

19. The summed NV East displacement mortality for autumn, mid-winter and spring using the SNCB recommended rates is 20 individuals (at 100% displaced and 10% mortality within the 4 km buffer) while the evidence-based mortality is 2 individuals (at 90% displacement and 1% mortality within the 4 km buffer, allowing for

rounding), although these figures include an unknown degree of double counting due to overlaps in the populations in each period.

20. It should be noted that the summed mortality for NV East within the 2 km buffer was 15 individuals (compared with 20 within the 4 km buffer) which further reduces the magnitude of predicted impact.

21. This additional mortality would increase the background mortality by a maximum of 0.66% which would be undetectable, although it is more likely there would be no effect at all. Therefore, during the entire nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of the highly precautionary assessment approach recommended by the SNCBs, and the additional precaution due to the potential for double counting. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

1.1.1.1.2 Norfolk Vanguard West

Autumn migration

22. Using the seasonal mean peak autumn migration abundance on NV West and 4 km buffer of 25, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- Three individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.5);
- No individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.5).

23. The BDMPS for red-throated diver in autumn is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of three individuals to this would increase the mortality rate by 0.01%, while the evidence-based prediction (0) results in no increase in mortality.

24. It should also be noted that the mean abundance within NV West and the 2 km buffer was 18 individuals (compared with 25 within the 4 km buffer) which would reduce the magnitude of impact by almost 30% to 2 individuals.

25. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.5 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard West (and 4 km buffer) during the autumn migration season that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	1
3	0	0	0	0	0	0	1	1	1	1
4	0	0	0	0	1	1	1	1	1	1
5	0	0	0	1	1	1	1	1	1	1
6	0	0	0	1	1	1	1	1	1	2
7	0	0	1	1	1	1	1	1	2	2
8	0	0	1	1	1	1	1	2	2	2
9	0	0	1	1	1	1	2	2	2	2
10	0	1	1	1	1	2	2	2	2	3
20	1	1	2	2	3	3	4	4	5	5
30	1	2	2	3	4	5	5	6	7	8
50	1	3	4	5	6	8	9	10	11	13
75	2	4	6	8	9	11	13	15	17	19
100	3	5	8	10	13	15	18	20	23	25

Mid-winter

26. Using the seasonal mean peak winter abundance on NV West and 4 km buffer of 356, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- 36 individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.6);
- Three individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.6).

27. The BDMPS for red-throated diver in winter is 10,177 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 2,320 (10,177 x 0.228). The addition of 36 individuals to this would increase the mortality rate by 1.5%, while the evidence-based prediction of 3 would increase the mortality rate by 0.13%.

28. It should be noted that the mean abundance within NV West and the 2 km buffer was 235 individuals (compared with 356 within the 4 km buffer) which would reduce the magnitude of predicted impact by 33% to 23 individuals using the SNCB recommended rates.

29. Using the highly precautionary SNCB approach this indicates the potential for a low magnitude of effect, while the evidence-based approach indicates a negligible magnitude of effect.

30. Therefore, during the winter period, since the species is of high sensitivity to disturbance, on the basis of the evidence based displacement and mortality rates the impact significance remains **minor adverse** as presented in the ES and **minor to moderate adverse** using the precautionary SNCB methods.

Table 1.6 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard West (and 4 km buffer) during the winter period that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	1	1	1	2	2	2	3	3	4
2	1	1	2	3	4	4	5	6	6	7
3	1	2	3	4	5	6	7	9	10	11
4	1	3	4	6	7	9	10	11	13	14
5	2	4	5	7	9	11	12	14	16	18
6	2	4	6	9	11	13	15	17	19	21
7	2	5	7	10	12	15	17	20	22	25
8	3	6	9	11	14	17	20	23	26	28
9	3	6	10	13	16	19	22	26	29	32
10	4	7	11	14	18	21	25	28	32	36
20	7	14	21	28	36	43	50	57	64	71
30	11	21	32	43	53	64	75	85	96	107
50	18	36	53	71	89	107	125	142	160	178
75	27	53	80	107	134	160	187	214	240	267
100	36	71	107	142	178	214	249	285	320	356

Spring migration

31. Using the seasonal mean peak autumn migration abundance on NV West and 4 km buffer of 197, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- 20 individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.7);
- Two individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.7).

32. The BDMPS for red-throated diver in spring is 13,277 (Furness 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number

of individuals expected to die is 3,027 (13,277 x 0.228). The addition of 20 individuals to this would increase the mortality rate by 0.66%, while the evidence-based prediction (2) results in an increase in mortality of 0.07%.

33. It should be noted that the mean abundance within NV West and the 2 km buffer was 127 individuals (compared with 197 within the 4 km buffer) which would reduce the magnitude of predicted impact by 35%, to 13 individuals using the SNCB recommended rates.

34. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.7 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard West (and 4 km buffer) during the spring migration period that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	1	1	1	1	1	2	2	2
2	0	1	1	2	2	2	3	3	4	4
3	1	1	2	2	3	4	4	5	5	6
4	1	2	2	3	4	5	6	6	7	8
5	1	2	3	4	5	6	7	8	9	10
6	1	2	4	5	6	7	8	9	11	12
7	1	3	4	6	7	8	10	11	12	14
8	2	3	5	6	8	9	11	13	14	16
9	2	4	5	7	9	11	12	14	16	18
10	2	4	6	8	10	12	14	16	18	20
20	4	8	12	16	20	24	28	32	35	39
30	6	12	18	24	30	35	41	47	53	59
50	10	20	30	39	49	59	69	79	89	99
75	15	30	44	59	74	89	103	118	133	148
100	20	39	59	79	99	118	138	158	177	197

Complete nonbreeding season

35. The summed NV West displacement mortality for autumn, mid-winter and spring using the SNCB recommended rates is 59 individuals (at 100% displaced and 10% mortality within the 4 km buffer) while the evidence-based mortality is 5 individuals (at 90% displacement and 1% mortality within the 4 km buffer), although these

figures include an unknown degree of double counting due to overlaps in the populations in each period.

36. It should be noted that the summed mortality for NV West within the 2 km buffer was 38 individuals (compared with 59 within the 4 km buffer) which further reduces the magnitude of predicted impact.

37. The maximum additional mortality would increase the background mortality rate by a maximum of 1.95%. Using the highly precautionary SNCB approach this indicates the potential for a low magnitude of effect, while the evidence-based approach indicates a negligible magnitude of effect.

38. Therefore, during the entire nonbreeding period, the magnitude of effect is assessed as low to negligible and as the species is of high sensitivity to disturbance, on the basis of the evidence based displacement and mortality rates the impact significance remains **minor adverse** as presented in the ES and **minor to moderate adverse** using the precautionary SNCB methods.

1.1.1.1.3 Norfolk Vanguard East and Norfolk Vanguard West

39. The worst case displacement impact has been assessed on the basis that both NV East and NV West would be completely developed, although this is highly precautionary since even if each site contains half the total number of turbines it is very unlikely they would be distributed across the entirety of both sites (i.e. the actual developed area would be less than the sum of the total area of both assumed by this approach).

40. The combined NV east and NV west assessment (Table 1.8) also presents the range of predicted outcomes with inclusion of uncertainty in the estimated population sizes as requested by Natural England. This uses the upper and lower 95% confidence intervals of abundance for the wind farm areas plus 4 km buffers (as presented in Vattenfall 2018, Offshore Ornithology Technical Appendix 13.1).

Table 1.8 Red-throated diver displacement mortality across all seasons for NV East, NV West (inc. 4 km buffers) and both combined, with full consideration for uncertainty in abundance estimates and assessed using the SNCB recommended rates and the evidence-based rates.

Period	Site	Displacement rate (%)	Mortality rate (%)	Displacement mortality including uncertainty in population size		
				Lower confidence interval	Mean abundance	Upper confidence interval
NV East	Autumn	100	10	0	5	11
		90	1	0	0	1
	Mid-winter	100	10	0	3	8
		90	1	0	0	1

Period	Site	Displacement rate (%)	Mortality rate (%)	Displacement mortality including uncertainty in population size		
				Lower confidence interval	Mean abundance	Upper confidence interval
	Spring	100	10	0	12	38
		90	1	0	1	3
	Total	100	10	0	20	59
		90	1	0	1	5
NV west	Autumn	100	10	0	3	8
		90	1	0	0	1
	Mid-winter	100	10	7	36	71
		90	1	1	3	6
	Spring	100	10	11	20	29
		90	1	1	2	3
	Total	100	10	18	59	108
		90	1	2	5	10
NV East & NV West	Total	100	10	18	79	167
		90	1	2	6	15

41. The combined displacement mortality across both NV East and NV West (inc. 4 km buffers) for the complete nonbreeding period using the mean abundance estimates would be 79 individuals (at 100% displaced and 10% mortality) while the evidence-based mortality is 6 individuals (at 90% displacement and 1% mortality). Both these sets of figures include a very large degree of precaution: in the approach to estimating numbers affected, the assumption that both sites will be fully developed and double counting across overlapping seasons. Furthermore, the evidence review has found that displacement is very likely to extend no further than 1.5 km beyond the wind farm. As discussed above this would reduce the number affected by around 33% (i.e. the upper mean estimate would be around 52, compared with 79). Given these existing sources of precaution, it is considered highly unrealistic to add further precaution to this assessment through the incorporation of predictions derived using the upper confidence intervals of abundance. Furthermore, any consideration of the upper estimates should be balanced against the lower confidence estimates which suggest a maximum mortality of 18 using the highly precautionary SNCB methods.

42. The various assessment approaches generate the following predicted increases in mortality for NV East and NV West combined:

- 2.6% (4 km buffer, 100% displacement, 10% mortality);
- 1.7% (2 km buffer, 100% displacement, 10% mortality);
- 0.2% (4 km buffer, 90% displacement, 1% mortality);
- 0.1% (2 km buffer, 90% displacement, 1% mortality).

43. Using the highly precautionary SNCB approach (detailed above), combined displacement across both NV East and NV West (4 km buffer, 100% displaced, 10% consequent mortality) would increase the background mortality by 2.6%, while the evidence-based approach indicates a maximum increase in background mortality of 0.2% (4 km buffer) and a more realistic increase of 0.1% (within the 2 km buffer).

44. Therefore, during the entire nonbreeding period across both NV East and NV West these therefore predict a low to negligible magnitude of effect and as the species is of high sensitivity to disturbance, on the basis of the evidence based displacement and mortality rates the impact significance remains **minor adverse** as presented in the ES and **minor to moderate adverse** using the precautionary SNCB methods.

1.1.1.2 Cumulative: Disturbance and displacement from offshore infrastructure

45. Cumulative red-throated diver displacement mortality has been estimated for wind farms in the south-west North Sea BDMPS (Furness 2015) which have the potential to contribute to a cumulative effect. This has been conducted using the precautionary rates of displacement and mortality recommended by the SNCBs (100% displacement and 10% mortality within the 4 km buffer) as well as those derived from the review of evidence for this species, reported in Annex 1 (90% displacement and 1% mortality).

46. The original cumulative assessment (Vattenfall 2018) focused on the core area for this species, as highlighted by the species' inclusion in the Greater Wash SPA and Outer Thames SPA. This included wind farms located between Triton Knoll and Thanet on the basis that the coastal waters in this region cover the primary over-wintering areas. In their review of the assessment, Natural England (2018) requested inclusion of all wind farms in the south-west North Sea BDMPS for red-throated diver (Furness 2015). Natural England (2018) also stated that none of the wind farms included in the cumulative assessment should be considered as part of the baseline (i.e. as if their effects were already accounted for in the Norfolk Vanguard survey data) because the BDMPS population estimates pre-dated all the wind farm construction dates. Hence, no distinction between wind farms has been made on the basis of date of commission.

47. A review of the impact assessments for the additional wind farms in the relevant BDMPS identified three categories with respect to red-throated divers: wind farms with no population estimates presented (Dogger Bank sites and Blyth demonstrator), coastal wind farms with low numbers of over-wintering birds reported (Teesside, Humber Gateway and Westermost Rough) and wind farms with sightings made during months considered to belong to the breeding season (Hornsea projects). A summary of the data for all these sites is presented along with those assessed in the ES (Table 1.9).

Table 1.9 Summary of red-throated diver assessments for older wind farms in south west North Sea BDMPS with potential to contribute to a cumulative operational displacement impact. Displacement estimates presented as the evidence-based combination (90%-1%) and SNCB guidance combination (100%-10%).

Wind farm	Red-throated diver displacement assessed?	Estimated no. of red-throated diver mortalities due to displacement	Conclusion for NV cumulative assessment
Scroby Sands	Not assessed	No number presented	NA
Kentish Flats	Yes: qualitative	No number presented	NA
Lynn & Inner Dowsing	Yes: qualitative	No number presented	NA
Gunfleet Sands	Yes: qualitative	‘very small'	NA
Thanet	Yes: quantitative	<1 - 2	Included
Sheringham Shoal	Not assessed	No number presented	NA
Greater Gabbard	Yes: quantitative	4 - 40	Included
London Array	Yes: qualitative	No number presented	NA
Lincs	Yes: qualitative	No number presented	NA
Kentish Flats Extension	Yes: qualitative	No number presented	NA
Galloper	Yes: quantitative	1 - 14	Included
Dudgeon	Not assessed	No number presented	NA
Race Bank	Not assessed	No number presented	NA
Triton Knoll	Not assessed	No number presented	NA
Dogger Bank Creyke Beck A & B	Not assessed	No number presented	NA
Dogger Bank Teesside A / Sofia	Not assessed	No number presented	NA
Blyth Demonstrator	Not assessed	No number presented	NA
Teesside	Not assessed	No number presented	NA
Westermost Rough	Not assessed	No number presented	NA
Humber Gateway	Not assessed	No number presented	NA
Hornsea 1	Not assessed	No number presented	NA
Hornsea 2	Not assessed	No number presented	NA
Hornsea 3	Not assessed	No number presented	NA

48. Although several of the additional wind farms recorded red-throated diver in low numbers in their surveys, none undertook assessment of displacement impacts. Thus, the inclusion of the additional wind farms has not altered the number of individuals assessed as at risk of displacement (although the tables present revised rates of displacement and consequent mortality). This is not wholly surprising since most of these additional wind farms are located in areas typically considered to be less preferred by species (i.e. farther offshore and in depths of >20 m). In total, for the wind farms outside the former East Anglia zone the displacement assessments indicated that between 6 and 56 individuals would be at risk of mortality. This total has been included in the cumulative assessment, together with the former East Anglia zone wind farms. Note that the latter also includes revised estimates for Thanet Extension, using the estimates presented in the ES.

Table 1.10 Red-throated diver cumulative displacement mortality calculated on the basis of the SNCB recommended rates of 100% displacement and 10% mortality and the evidence-based rates of 90% displacement and 1% mortality (all within the 4 km buffer).

Wind farm	Autumn	Midwinter	Spring	Annual
Older projects (see Table 13.65)	N/A	N/A	N/A	6 - 56
Thanet Extension	0	2 - 19	0 - 4	2 - 23
East Anglia ONE	0.4 - 5	1 - 10	1.4 - 15	3 -30
East Anglia THREE	0.4 - 5	0.2 – 2	2 - 20	0.6 - 27
Norfolk Vanguard East	0.4 – 5	0.2 – 3	1 – 12	1.6 - 20
Norfolk Vanguard West	0 – 3	3 - 36	2 - 20	5 - 59
Total (rounded)	1 - 18	6 - 70	6 - 71	18 - 215

49. The estimated cumulative red-throated diver mortality for all wind farms in the south west North Sea BDMPS region is between 18 and 215, on the basis that all individuals within 4 km of the wind farms are displaced. If it is assumed that the abundance at the other wind farms within 2 km is 33% lower than within 4 km (as is the case at Norfolk Vanguard), then application of the evidence-based finding that displacement extends no further than 1.5 km (Annex 1), these totals would decline to between 12 and 142.

50. The largest BDMPS for red-throated diver is 13,277 (Furness 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of between 18 and 215 to this would increase the mortality rate by 0.6% to 7% (or 0.4% to 4.7% within 2 km).

51. The biogeographic population for red-throated diver is 27,000 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 the number of individuals expected to die is 6,156 (27,000 x 0.228). The addition of between 18

and 215 would increase the mortality rate by to 0.3% to 3.5% (or 0.2% to 2.3% within 2 km).

52. To summarise, the various assessment approaches generate the following predicted increases in cumulative mortality for the BDMPS population:

- 7% (4 km buffer, 100% displacement, 10% mortality);
- 4.7% (2 km buffer, 100% displacement, 10% mortality);
- 0.6% (4 km buffer, 90% displacement, 1% mortality);
- 0.4% (2 km buffer, 90% displacement, 1% mortality);

53. The following predicted increases in cumulative mortality for the biogeographic population:

- 3.5% (4 km buffer, 100% displacement, 10% mortality);
- 2.3% (2 km buffer, 100% displacement, 10% mortality);
- 0.3% (4 km buffer, 90% displacement, 1% mortality);
- 0.2% (2 km buffer, 90% displacement, 1% mortality).

54. As discussed in preceding sections, the mortality total combines multiple sources of precaution:

- The evidence review found that 90% displacement and 1% mortality are more appropriate than the 100% and 10% recommended by the SNCBs;
- Each wind farm assessment has assumed that all birds within 4 km of the wind farm lease boundary are potentially affected, whereas the evidence suggests displacement declines with distance from wind farm boundaries and in some cases has been reported as zero by 2 km;
- It includes an unknown degree of double counting across seasons since some individuals will be present within more than one season and could also potentially move between these sites;
- The Norfolk Vanguard East 4 km buffer includes part of the East Anglia THREE wind farm and 4 km buffer and vice versa so including the buffer for both sites leads to double counting birds in the overlapping area; and
- One third of the total is predicted to occur during the spring migration period when the potential consequences of displacement are expected to be much lower since most individuals are on migration and passing through at this time.

55. Furthermore, the method used for assessing displacement impacts has no means to explicitly incorporate wind farm design modifications, specifically with respect to turbine spacing. Most wind farms are constructed with fewer, larger diameter turbines than specified in their consents. Due to the need to minimise the turbulence downwind from a turbine, as turbine rotor diameter increases, so the spacing between turbines increases (since the wake effect is a function of rotor

diameter). Since the underlying assumption for displacement from operational wind farms is that birds avoid the turbines themselves, it follows logically that as turbine spacing increases so the stimulus for avoidance behaviour decreases, thereby permitting more individuals to enter a wind farm. This is relevant because the displacement assessments for other wind farms represent the predictions for the consented designs not the final ones utilising fewer larger turbines. Thus, in addition to the sources of precaution listed above, there also needs to be allowance for the reduced displacement from built wind farms compared with the consented versions.

56. For example, East Anglia ONE was originally assessed on the basis of 333 turbines, reduced to 240 for consent with the final design further reduced to 102 turbines with only a small decrease in project area. Thus, the final wind farm will have less than one third the original number of proposed (and assessed) turbines. This will almost certainly reduce the magnitude of displacement. The total also includes an unrealistic worst case scenario for Norfolk Vanguard with complete displacement from both NV East and NV West, calculated as the summed total effect for both sites. This corresponds to two times the actual maximum number of turbines which could be installed across both sites, since the full assessments for NV East and NV West assume all turbines are located in just one of the sites, with no development in the other site. In reality, it is more reasonable to assume that combined displacement would lie between the values obtained for NV East and NV West (e.g. at the SNCB recommended 100% displacement and 10% mortality (within the 4 km buffer), rather than the sum total of 79 (or 6, at 90%-1%), the mortality would be between 20 and 59).

57. To inform the assessment, the potential effect on the background mortality resulting from combinations of displacement and mortality have been calculated for the BDMPS population and for the biogeographic population (Table 1.11 and Table 1.12). The colour shading in these tables indicates the displacement and mortality combinations which result in background mortality increases of less than 1%, between 1% and 2% and between 2% and 3%. For the BDMPs population, at all displacement rates combined with 1% consequent mortality result in background mortality rising by less than 1% (i.e. below the level at which this effect could be detected).

Table 1.11 Red-throated diver cumulative displacement matrix. Levels of mortality which would increase the baseline mortality of the smaller BDMPS population by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	2	4	6	9	11	13	15	17	19	22
2	4	9	13	17	22	26	30	34	39	43
3	6	13	19	26	32	39	45	52	58	65
4	9	17	26	34	43	52	60	69	77	86
5	11	22	32	43	54	65	75	86	97	108
6	13	26	39	52	65	77	90	103	116	129
7	15	30	45	60	75	90	105	120	135	151
8	17	34	52	69	86	103	120	138	155	172
9	19	39	58	77	97	116	135	155	174	194
10	22	43	65	86	108	129	151	172	194	215

58. Against the larger biogeographic population, 100% displacement would have an undetectable effect (i.e. increase the background rate by less than 1%) when combined with up to 2% mortality, while at displacement rate of 80% and 90% the same applies up to 3% mortality.

Table 1.12 Red-throated diver cumulative displacement matrix. Levels of mortality which would increase the baseline mortality of the biogeographic population by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	2	4	6	9	11	13	15	17	19	22
2	4	9	13	17	22	26	30	34	39	43
3	6	13	19	26	32	39	45	52	58	65
4	9	17	26	34	43	52	60	69	77	86
5	11	22	32	43	54	65	75	86	97	108
6	13	26	39	52	65	77	90	103	116	129
7	15	30	45	60	75	90	105	120	135	151
8	17	34	52	69	86	103	120	138	155	172
9	19	39	58	77	97	116	135	155	174	194
10	22	43	65	86	108	129	151	172	194	215

59. On the basis of the worst case SNCB approach the cumulative red-throated diver operational displacement impact magnitude is assessed as low. Therefore, as the species is of high sensitivity to disturbance, the cumulative impact significance would be **moderate adverse**.

60. On the basis of the evidence review (Annex 1) it is considered that the most realistic (and still precautionary) combination of displacement and consequent mortality rates is 90% and 1%, respectively operating within no more than 2 km of the wind farm boundary. On this basis the cumulative red-throated diver operational displacement impact magnitude is assessed as negligible. Therefore, as the species is of high sensitivity to disturbance, the cumulative impact significance would be **minor adverse**.

1.1.2 Potential impacts during construction

Offshore Export Cable Installation

61. The magnitude of disturbance to red-throated diver from construction vessels has been estimated on a worst case basis. This assumes that there would be 100% displacement of birds within a 2 km buffer surrounding the source, in this case around a maximum of two cable laying vessels. This 100% displacement from vessels is consistent with Garthe and Hüppop (2004) and Schwemmer *et al.,* (2011) since they suggested that all red-throated divers present fly away from approaching vessels at a distance of often more than 1 km.

62. In order to calculate the number of red-throated divers that would potentially be at risk of displacement from the offshore cable corridor during the cable laying process, the density of red-throated divers in the SPA along the section crossed by the offshore cable corridor was estimated. This was derived from a review of the Greater Wash SPA proposal details (Natural England and JNCC, 2016). This indicated that the peak density of birds in the SPA crossed by the cable route was between 1.36 and 3.38 per km^2.

63. The worst case area from which birds could be displaced was defined as a circle with a 2 km radius around each cable laying vessel, which is 25.2 km^2 (2 x 12.6 km^2). If 100% displacement is assumed to occur within this area, then a peak of between 34 and 85 divers could be displaced at any given time. This would lead to a 1 to 1.5% increase in diver density in the remaining areas of the SPA assuming that displaced birds all remain within the SPA. As the vessels move it is assumed that displaced birds return and therefore any individual will be subjected to a brief period of impact. It is considered reasonable to assume that birds will return following passage of the vessel since the cable laying vessels will move at a maximum speed of 400 m per hour if surface laying, 300 m per hour for ploughing and 80 m per hour if trenching (Chapter 5 Project Description). This represents a maximum speed of 7 m per minute. For context, a modest tidal flow rate for the region would be in the region of 1 m per second (60 m per minute). The tide would therefore be flowing about nine times faster than the cable laying vessel. Consequently, for the purposes of this assessment it can be assumed that the estimated number displaced by a cable

laying vessel represents the total number displaced for the duration of cable laying operations. The export cable may be installed in a single phase or two phases, therefore this impact could occur in a maximum of two winters. However, the total duration is the same in each case (single phase of 6 months or two phases of 3 months, separated by up to three years) and therefore the magnitude of impact is the same for both options.

64. In the ES, the potential consequence of displacement from construction vessels was based on a mortality rate of 5% for displaced individuals. NE requested that this should be increased to 10% (although note that the review in Annex 1 does not support such use of such a precautionary rate).

65. At the 5% mortality rate it was predicted that a maximum of between 2 to 4 birds would be expected to die across the equivalent of one entire winter period (September to April, although note this could occur in a single phase of 6 months or two phases of 3 months) as a result of any potential displacement effects from the offshore cable installation activities, which would be restricted to the equivalent of a single season, and only if cable laying takes place during these months. This would be doubled (4 to 8 birds) at the NE recommended rate of 10%. Even when compared to the smaller winter BDMPS for this species (10,177; Furness, 2015) it is clear that this highly precautionary assessment will generate an effect of negligible magnitude.

66. The construction works, specifically offshore cable laying, are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Norfolk Vanguard East

67. There is potential for disturbance and displacement of red-throated divers due to construction activity and associated vessel traffic. However, construction will not occur across the whole of the proposed wind turbine array area simultaneously or every day but will be phased with a maximum of two foundations expected to be installed simultaneously. Consequently, the effects will occur only in the areas where vessels are operating at any given point and not the entire NV East site.

68. For the ES a precautionary mortality rate of 5% mortality for individuals displaced by construction vessels was used. However, NE requested that a more precautionary morality rate of 10% should be used, and the following provides an update to the original assessment with also this mortality rate used.

69. During **autumn migration**, with a seasonal peak density of 0.09/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 2

individuals (0.09 x 12.56 x 2) could be at risk of displacement and up to 0.1 at risk of mortality (at 5%) and 0.2 (at 10%) in a maximum of two autumn periods.

70. At the average baseline mortality rate for red-throated diver of 0.228 the number of individuals expected to die in the autumn BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 0.2 to this increases the mortality rate by 0.006%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

71. During **winter**, with a seasonal peak density of 0.06/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 2 individuals (0.06 x 12.56 x 2) could be at risk of displacement and up to 0.1 at risk of mortality (at 5%) and 0.2 at 10%) during a maximum of two winter periods.

72. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the winter BDMPS is 2,320 (10,177 x 0.228). The addition of a maximum of 0.2 to this increases the mortality rate by 0.008%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

73. During **spring**, with a seasonal peak density of 0.26/km^2 and a precautionary 2km radius of disturbance around each construction vessel, 7 individuals (0.26 x 12.56 x 2) could be at risk of displacement and up to 0.3 at risk of mortality (at 5%) and 0.6 (at 10%) during a maximum of two spring periods.

74. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the spring BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 0.6 to this increases the mortality rate by 0.02%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

75. The combined nonbreeding impact of construction, with approximately 0.5 individuals (at 5%) or 1 (at 10%) at risk of construction displacement mortality, will

be similarly undetectable against background levels (this would increase the background mortality of the smallest BDMPS population by a maximum of 0.04%). Therefore, during the combined nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Norfolk Vanguard West

76. There is potential for disturbance and displacement of red-throated divers due to construction activity, including wind turbine construction and associated vessel traffic. However, construction will not occur across the whole of the proposed wind turbine array area simultaneously or every day but will be phased with a maximum of two foundations expected to be installed simultaneously. Consequently, the effects will occur only in the areas where vessels are operating at any given point and not the entire NV West site.

77. For the ES a precautionary mortality rate of 5% mortality for individuals displaced by construction vessels was used. However, NE requested that a more precautionary morality rate of 10% should be used, and the following provides an update to the original assessment with the addition of this mortality rate.

78. During **autumn migration**, with a seasonal peak density of 0.01/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, less than 1 individual (0.01 x 12.56 x 2 = 0.25) could be at risk of displacement and up to 0.01 (at 5%) and 0.02 (at 10%) at risk of mortality during a maximum of two autumn periods.

79. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the autumn BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 0.02 to this increases the mortality rate by 0.006%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

80. During **winter**, with a seasonal peak density of 0.48/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 12 individuals (0.48 x 12.56 x 2) could be at risk of displacement and up to 0.6 (at 5%) and 1.2 (at 10%) at risk of mortality during a maximum of two winter periods.

81. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the winter BDMPS is 2,320 (10,177 x 0.228). The addition of a maximum of 1.2 to this increases the mortality rate by 0.06%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

82. During spring, with a seasonal peak density of 0.37/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 9 individuals (0.37 x 12.56 x 2) could be at risk of displacement and up to 0.5 (at 5%) and 1 (at 10%) at risk of mortality during a maximum of two spring periods.

83. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the spring BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 1 to this increases the mortality rate by 0.04%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

84. The combined nonbreeding impact of construction, with approximately 2.2 individuals at risk of construction displacement mortality, will be similarly undetectable against background levels (this would increase the background mortality of the smallest BDMPS population by 0.1%). Therefore, during the combined nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Norfolk Vanguard East and Norfolk Vanguard West

85. Although construction may occur on both NV East and NV West at the same time, the maximum number of simultaneous piling events would remain two. Therefore, the combined impact across the two sites would not exceed that assessed for the two sites alone and no further assessment is required.

1.1.3 Operational vessel movements

86. Vessel movements during the operation of the wind farm for maintenance activities have the potential to disturb red-throated divers. However, within the confines of

the wind farm sites and the 4 km buffer, the magnitude of displacement due to the wind farm itself (assessed as 90-100%) is such that there would be virtually no additional effect caused by vessel movements (i.e. almost all individuals will already have been displaced). Therefore, no further assessment for operational vessel movements within the wind farm sites (and buffers) is required.

87. The operation and maintenance port has not been confirmed at this stage. However, it is clear from consideration of the existing volume of shipping traffic through the region (Shipping and Navigation assessment, Appendix 15.01 vol 3, figures 15.1 and 15.2), which includes the Greater Wash SPA and Outer Thames Estuary SPA, that the addition of vessels transiting to and from the port and the wind farm (approx. 1.2 vessel movements per day) will have a negligible effect on the levels of shipping disturbance over and above the average of almost 100 vessel movements per day (derived from AIS data, and therefore not including smaller vessels).

2 ANNEX 1. RED-THROATED DIVER DISPLACEMENT AND CONSEQUENT MORTALITY: ASSESSMENT OF EVIDENCE

Are red-throated divers displaced from operational offshore wind farms?

88. Yes. Dierschke et al. (2016) reviewed studies published up to 2016 that compared seabird abundances within offshore wind farms post-construction with baseline data from before construction. Studies at ten different offshore wind farms found 'strong avoidance' by red-throated divers or 'divers' (a category that was predominantly red-throated divers but included some black-throated divers) at nine of these sites, and 'weak avoidance' at one site. These results imply consistent displacement from offshore wind farms.

89. Skov et al. (2018) reported the results of the ORJIP bird avoidance study at Thanet offshore wind farm. They obtained 82 radar tracks and 42 laser rangefinder tracks of red-throated divers, which would appear to provide an adequate sample size to assess macro-avoidance of that wind farm, but they do not report on the avoidance behaviour of that species as it was not one of the key species in that study. Percival and Ford (2018) stated that there was clear evidence that red-throated divers strongly avoid Kentish Flats Extension and a 0-500m zone around that wind farm. Mendel et al. (2019) used aerial survey data in a BACI design for an area of the German Bight that included hot-spots for red-throated divers (including a Special Protection Area designated with nonbreeding red-throated diver as the feature) and four offshore wind farms. Red-throated diver density decreased highly significantly within and around the wind farms post-construction. In contrast however, APEM (2016) found clear evidence of displacement of red-throated divers by London Array offshore wind farm during construction, but no clear evidence of displacement during operation. Similarly, Gill et al. (2018) found statistically significant displacement of red-throated divers from Greater Gabbard offshore wind farm during construction, but no statistically significant difference between pre-construction and post-construction, although the trend was in the direction of a displacement effect similar to that reported in other studies.

How strong is the displacement effect?

90. Petersen et al. (2006) estimated that displacement of red-throated divers occurred up to at least 2 km from the outer turbines at Horns Rev, with a decline of about 90% in diver numbers within the operational wind farm and a similar decline in the 2 km buffer around the outer turbines but no significant change beyond 2 km. Fox and Petersen (2006) provide a methodological framework for assessing the degree of habitat loss caused by displacement, and suggest that divers were significantly affected within 2 km of Horns Rev.

91. Percival (2013) reported an 82% decline in red-throated diver density within Thanet wind farm but no significant effect outside the wind farm. Percival (2014) concluded that displacement of red-throated divers by Kentish Flats offshore wind farm was not evident more than 1 km from the outer turbines.

92. Welcker and Nehls (2016) reported a reduction of 90% in red-throated diver density within Alpha Ventus offshore wind farm during construction and the first three years of operation compared with baseline numbers before construction. Diver abundance reached an asymptote at a distance of about 1.5 km from the outer turbines, suggesting no displacement effect beyond 1.5 km.

93. Peterson et al. (2014) noted red-throated diver densities at Horns Rev II were lower than pre-construction densities up to 13 km from the wind farm, but they considered that the difference was likely to be due to other factors, and suggested that displacement by the wind farm may occur up to about 5-6 km from the outer turbines. Diver density within 4 km of the wind farm was reduced by about 25%.

94. At Kentish Flats extension, the overall reduction in density was 89% within the wind farm and 70% in the 0-500 m buffer. However, there was no statistically significant effect detectable beyond 500 m from the outer wind turbines (Percival and Ford 2018).

95. Mendel et al. (2019) found overall a 94% reduction in red-throated diver abundance within 3 km of operational wind farms in the German Bight compared with the pre-construction baseline. When they separated displacement effects of ship traffic from those of offshore wind farm structures, they found a 71% reduction in red-throated diver densities in the area within 3 km of the operational offshore wind farms attributable to the offshore wind farm structure, with an additional displacement of at least a further 14% attributable to associated ship traffic. Mendel et al. (2019) found that statistically significant avoidance could be detected up to 12 km from the wind farms, although beyond 3 km the displacement was at much lower levels than closer to the wind farm.

96. APEM (2016) found a decrease in diver density up to at least 10 km from the wind farm during construction, whereas during operation the density was similar to that pre-construction and higher than pre-construction at 2 km from the wind farm, suggesting a displacement distance of <1.5 km.

97. Gill et al. (2018) estimated displacement of 75% from Greater Gabbard offshore wind farm during post-construction but the change was not statistically significant due to high variance in the data. However, their distribution maps indicate that there

was little or no displacement of divers from the immediate area surrounding the wind farm.

98. Whereas most studies consistently found a marked decrease in red-throated diver densities within operational wind farms when compared to pre-construction data (Table 2.1), the distance outside the wind farm over which diver densities were reduced varied greatly among sites (Table 2.1). At the extremes, Percival (2013) found no reduction in diver density outside Thanet offshore wind farm even within 500 m of the outer turbines, whereas Mendel et al. (2019) found a statistically detectable reduction in density up to 12 km from the outer turbines. This variation is unexplained. However, it might relate to ecological conditions or to the seascape/landscape of the site. Behaviour may vary seasonally, for example, depending on ecological constraints at different times of year, such as may arise during flight-feather moult when birds may become flightless. Birds might show greater avoidance distances where they are unconstrained. At sites where suitable or optimal habitat is limited, birds might show lower displacement distances because of constraints imposed by habitat availability. Alternatively, divers may show stronger avoidance of visible structures at sea where these are against an 'empty' background seascape. Where structures are in front of a cluttered background of coast, perhaps especially a coast with industrial development, the offshore turbines may appear less prominent and/or may be seen by divers as less threatening. The largest distances from offshore wind farms over which diver densities were reduced were in the German Bight, a very large area of open sea far from the coast. The smallest displacement distances from offshore wind farms were at sites close to the UK coast where anthropogenic influences on the coastal scenery are high (Thanet, Kentish Flats).

Table 2.1 Summary of distances from offshore wind farms over which displacement of red-throated divers was assessed to occur, and the percentage reduction in red-throated diver densities within operational wind farms by comparison with baseline pre-construction densities.

Wind farm	Distance from outer turbines over which diver density was significantly reduced (km)	Percentage reduction in diver density within wind farm area	Reference
Thanet	0.0	82	Percival 2013
Kentish Flats Extension	0.5	89	Percival and Ford 2018
Greater Gabbard	<1.0	(75)*	Gill et al. 2018
Kentish Flats	1.0	-	Percival 2014
Gunfleet Sands	1.0	-	Barker 2011
London Array	<1.5	<50	APEM 2016
Alpha Ventus	1.5	90	Welcker & Nehls 2016
Horns Rev 1	2.0	90	Petersen et al. 2006
North Hoyle	2.5	-	May 2008

Wind farm	Distance from outer turbines over which diver density was significantly reduced (km)	Percentage reduction in diver density within wind farm area	Reference
Lincs	2-6	-	Webb et al. 2015
Horns Rev 2	5.5	50	Petersen et al. 2014
Butendiek, Amrumbank, Nordsee Ost, Meerwind Süd/Ost, Dan Tysk	12.0	94	Mendel et al. 2019

*But not statistically significant due to high variance in data so a tentative estimate

99. To conclude this section, it seems reasonable to assume that 90% of red-throated divers will be displaced from offshore wind farm sites. It is less clear how many will be displaced from waters surrounding offshore wind farms, as this seems to be highly variable among sites. For sites in UK waters it seems that an appropriate precautionary estimate would be that 90% of red-throated divers will be displaced from a buffer zone of 1.5 km from an offshore wind farm, although site-specific data from UK suggest that displacement of birds may only extend as far as 500 m beyond the outer turbines at least in some cases (Table 2.1).

Is there evidence for habituation of divers to offshore wind farms?

100. If red-throated divers habituate over time to the presence of offshore wind farms, then habitat loss might be negligible in the long term. Leopold and Verdaat (2018) found some evidence for auks habituating to Luchterduinen offshore wind farm and suggested a methodology to assess the extent of such habituation. However, Petersen and Fox (2007) suggested that there was no evidence of habituation of red-throated divers at Horns Rev, at least in the first few years of operation. Percival (2010) suggested that red-throated divers were starting to habituate to Kentish Flats offshore wind farm, with increasing numbers entering the wind farm in 2009-10. However, subsequent data suggest no habituation at all at this site (Percival 2014). Mendel et al. (2019) found no evidence of any habituation of red-throated divers to offshore wind farms in the German Bight. From this we conclude that currently there does not seem to be any clear evidence of red-throated divers habituating to offshore wind farms.

What are the likely consequences of displacement for individuals?

101. Consequences of displacement of red-throated divers have been summarised by Dierschke et al. (2017). They state that displacement could influence individual divers if offshore wind farm barrier effects or habitat loss result in a change in the bird’s energy budget. Under some circumstances, though not all, displacement could increase energy costs, or could result in decreased energy intake. The former could arise if birds had to fly more to avoid offshore wind farms or to reach more distant

foraging areas. The latter could arise if displacement was to lower quality habitat where food capture rates were lower, or if displacement resulted in an increase in diver density with a consequent increase in intra-specific competition. Alternatively, displacement may have no effect on individuals if birds are displaced into equally good habitat so that their energy budget is unaffected, or if birds could buffer any impact on energy budget by adjusting their time budget (for example by spending a higher proportion of the time foraging rather than resting in order to compensate for an increase in energy budget). According to Dierschke et al. (2017) 'red-throated divers appear capable of utilising a range of marine habitats and prey species. They also tend to occur at relatively low densities and not in large aggregations. Consequently, reduced prey intake caused by increased density-dependent competition or interference would seem unlikely. Red-throated divers are highly mobile in winter which may mean that they are able to find alternative foraging sites following displacement.' Red-throated divers are mostly widely dispersed during the nonbreeding season, with fewer than 4 birds per km^2 (Dierschke et al. 2017), which makes it difficult to imagine that density-dependent competition for food could apply in this species during the nonbreeding season, as birds at such low densities would not be able to deplete their food resource (small fish such as sprats, young herring, sandeels; Guse et al. 2009, Dierschke et al. 2017). Dierschke et al. (2017) also state 'year-round energetic budgets are unknown but this information is key to understanding the possible consequences of displacement… if red-throated divers tend to be in poorer condition in the nonbreeding season when displacement is occurring, displacement could have an impact on survival and productivity. However, if individuals are in relatively good condition during the nonbreeding season and spend only a small proportion of their daily activity budget foraging, they may have the capacity to buffer against any additional energetic expense of displacement and barrier effects'. Dierschke et al. (2017) also noted that red-throated divers use staging areas during migration between breeding and wintering grounds and spend several weeks undergoing a post-breeding moult during which they are flightless. Displacement at moulting grounds or staging areas may have different consequences from displacement in wintering areas.

102. In the context of overwinter survival, it is relevant that in many seabird species, most mortality occurs during winter (e.g. Coulson et al. 1983). This may be caused by a variety of factors, such as winter storms (Anker-Nilssen et al. 2018). However, there is also evidence that seabirds tend to be heavier in winter than during the breeding season (e.g. Coulson et al. 1983). It is inferred from this that most seabirds have relatively little difficulty in finding enough food during the nonbreeding season so can achieve higher body condition that buffers against short periods of adverse weather conditions. For example, puffins are 20-30% heavier in winter than in

summer as a result of storing fat during the nonbreeding season, and the same is true of guillemots (Anker-Nilssen et al. 2018). If the same pattern occurs in red-throated divers, which seems likely, an implication is that their body condition would not be greatly affected by plausible levels of displacement or disturbance, since their time budgets may not be constrained during this period.

103. For red-throated diver, Horswill and Robinson (2015) recommend use of baseline age of first breeding at 3 years old, adult (third year and older) survival 0.84, immature (second year) survival 0.62 and juvenile (first year) survival 0.6. The lower survival rates of juveniles suggests that if there were impacts of displacement from offshore wind farms then those might be most likely to arise among juvenile birds rather than adults.

104. The annual mortality of adult red-throated divers is around 16% per annum and this will include mortality (if any) caused by human disturbance in marine environments that has been occurring in previous decades. The amount of general ship traffic has increased up to the present time, but has been high since the 1950s (IMO, Oskin 2014), while numbers of fishing vessels increased during the early 20th century but have decreased slightly in recent decades (Uberoi 2017). It is known that red-throated divers often tend to fly off when an approaching ship is about 1-2km away (Schwemmer et al. 2011). There is a case to be made that the net energy costs of flying away from approaching ships (and consequent loss of foraging time and opportunity) is likely to be considerably greater than the energy cost of avoiding static structures such as offshore wind turbines. Given that all offshore wind farms in UK North Sea waters combined represent an extremely small fraction of potential foraging habitat of red-throated divers within UK North Sea waters, it would seem appropriate to assess the plausible additional mortality caused by offshore wind farm displacement, barrier effects and associated increases in ship traffic (both during construction and operation) as also being extremely small in relation to the existing total annual mortality (also given that this total annual mortality already includes any impact of existing (baseline) ship disturbance impacts: in 2012 an average of 86 vessel transits were identified by Automated Identification System data per day in the waters off East Anglia[1]; MMO 2014). In that context, to suggest that displacement from an offshore wind farm might add 5% or more to the baseline mortality for all individuals that are displaced seems inconsistent with a total annual mortality of red-throated diver adults of only 16%. Furthermore, this magnitude of effect appears even less likely when it is considered that the baseline mortality already includes the impacts from existing shipping activities, which almost certainly

[1] Note these data excluded commercial vessels less than 300 tonnes, recreational vessels, fishing vessels and military and government vessels on deployment.

cause disturbance to many red-throated divers many times per nonbreeding season. For example, Jarrett et al. (2018) reported that red-throated divers they watched in Orkney during the nonbreeding period (observations being of 1 to 3 individuals at a time in areas where vessel activity was likely) were subject to disturbance by vessels in 3 out of 30 five-minute observation periods. Out of a total of 7 disturbance incidents, birds flew away from the approaching vessel in 3 cases and swam away or dived in 4 cases. Such observations strongly suggest that disturbance by ships is likely to be of a much greater magnitude than displacement by offshore wind farms, and yet the impact of historical disturbance by ships must already be incorporated in the existing estimate of survival.

105. To set this in context, Goss-Custard et al. (2006a) assessed the impact of human disturbance of overwintering oystercatchers on mudflats and concluded that in winters with good feeding conditions, oystercatchers could be disturbed up to 1.5 times per hour before there was a reduction in their survival, whereas in winters with poor feeding conditions, being disturbed more than 0.5 times per hour resulted in an increased mortality risk. Madsen (1995) showed that pink-footed geese that were subject to high levels of human disturbance causing them to disperse from a spring staging area had lower breeding success than geese that were not subject to disturbance. By contrast, red-throated divers are almost certainly subject to multiple instances of disturbance due to vessel movements each winter and these evidently add up to less than a 16% per annum mortality (and obviously much less since much of the 16% per annum must be due to a wide range of natural mortality factors).

What are the likely consequences of displacement for the population?

106. Sutherland (1996) and Newton (1998) pointed out that for migrant birds, such as red-throated divers, population change following habitat loss in their nonbreeding area would depend on the relative strength of density-dependence in the breeding area and in the wintering area. If the population was regulated by density-dependent competition for breeding resources then habitat loss in the nonbreeding area may be unimportant. Goss-Custard et al. (1997) also pointed out that nonbreeding season habitat loss would only result in a decrease in a waterbird population if the population was subject to density-dependent competition for resources and population size was at carrying capacity of the environment.

107. Evidence strongly indicates that red-throated divers are limited by competition for safe breeding sites within range of foraging waters (Merrie 1978, Nummi et al. 2013, Rizzolo et al. 2014, Dahlen and Eriksson 2016), but they are probably not in competition for resources during the nonbreeding season (Dierschke et al. 2012, 2017). This would suggest that their population size will be limited by breeding habitat suitability and not by wintering habitat (Newton 1998). Loss of wintering

habitat would, therefore, have little or no impact on red-throated diver numbers unless habitat loss was so extensive that nonbreeding season habitat became a limiting factor for the population because their density increased so much that interference competition or prey depletion became a driving factor.

108. Topping and Petersen (2011) used an Individual Based Model to assess cumulative impact of displacement by offshore wind farms for the nonbreeding red-throated diver population migrating to Danish and Baltic waters. Their model made many assumptions and simplifications with very limited supporting evidence, but concluded that any population level cumulative impact of all operational and proposed offshore wind farms in Danish waters was likely to reduce diver numbers in the Baltic flyway by about 0.1%, whereas the cumulative impact of all operational and proposed offshore wind farms in Baltic waters might reduce numbers by 1.7% (Danish Energy Agency 2013).

109. Probably the most likely consequence is that displacement of red-throated divers will have effects which are too small to detect, as they are unlikely to be subject to density-dependent competition for resources during the nonbreeding season (Dierschke et al. 2017). Even though there are now many offshore wind farms in the southern North Sea and in the Baltic, the total area of these represents a very small fraction of the habitat used by nonbreeding red-throated divers throughout the southern North Sea and Baltic, so that cumulative habitat loss for red-throated divers is very small. The increase in density of red-throated divers caused by displacement away from offshore wind farms will therefore be extremely slight at the regional or biogeographic scale. However, the proportion of habitat lost may be much higher over certain small areas. For example, Mendel et al. (2019) estimated that displacement from offshore wind farms in the German Bight results in the effective loss of 8.8% of the Eastern German Bight SPA habitat for these birds. However, it is important to note that while the SPA boundary reflects historical distributions of red-throated divers, it does not necessarily follow that this represents the actual extent of suitable habitat in the area. So, displacement may move a proportion of birds out of the SPA, but this does not necessarily mean they will no longer be able to forage successfully.

110. Displacement from offshore wind farms must also be considered alongside disturbance caused by ships, as red-throated divers tend to fly away from approaching ships and that will increase their energy budget. Disturbance by ships seems likely to have the potential to affect red-throated diver energy budgets more than displacement from offshore wind farms, but these two effects may also interact. For example, birds displaced from offshore wind farms might move into habitat with higher levels of ship traffic.

111. To set the red-throated diver example in context, it is likely that red-throated divers do not experience prey depletion or interference competition while foraging (Dierschke et al. 2017). This means they are unlikely to be subject to density-dependent effects that would increase mortality if habitat loss (displacement) resulted in an increase in density in remaining areas. At the other extreme, it is known that many shorebirds that feed on mudflats are subject to strong interference competition and prey depletion (Goss-Custard et al. 2006b). These effects reflect the much more limited resource availability for waders feeding on inter-tidal areas, both spatially and temporally and the very much higher density of waders.

112. Estuarine habitat loss caused by barrages at Cardiff Bay and Rhymney resulted in an increase in mortality of 3.17% of displaced redshanks, a species known to be subject to strong density-dependent competition for food in winter due to both prey depletion and interference (Goss-Custard et al. 2006b). Oystercatchers are also known to be strongly susceptible to interference competition in winter on tidal mudflats. At Oosterschelde, Netherlands, two-thirds of the tidal mudflat area was destroyed by coastal engineering works (the Delta Works). There was no difference in oystercatcher winter adult survival or in movement rates before and after this habitat loss, although survival was reduced in severely cold winters compared to mild winters (Duriez et al. 2009). A study of the consequences of saltmarsh habitat loss for individually colour marked dark-bellied Brent geese followed the fate of displaced geese for 13 years after loss of saltmarsh habitat (Ganter et al. 1997). Displaced birds moved more often to less preferred sites that were not filled to capacity than did control birds. However, no significant differences in subsequent survival or fecundity of displaced birds could be found compared to control birds, although there may have been a slight but not statistically significant trend towards displaced birds performing less well than controls (Ganter et al. 1997). The researchers concluded that 'if alternative sites are available there may be no significant fitness consequences to forced dispersal' (i.e. displacement).

113. Based on our understanding of their winter feeding ecology and susceptibility to density-dependent competition, any effect of displacement of red-throated divers would be expected to be much less than seen in redshanks, and would be unlikely to be greater than seen in oystercatchers or dark-bellied Brent geese.

114. Despite the uncertainty about impacts on nonbreeding red-throated divers, the available evidence suggests that the most likely result is that there will be little or no impact on adult survival, and that any impact would probably be undetectable at the population level. A tracking study is currently underway which should provide the first estimates of the time-activity budgets of nonbreeding red-throated divers, and on their movements within winters (O'Brien et al. 2018). This should enable stronger

conclusions to be reached about the potential consequences of displacement for this species.

115. To conclude this section, we must acknowledge that the impact of displacement of red-throated divers by offshore wind farms is unknown. However, we do know that natural mortality of adult red-throated divers (including impacts of disturbance and displacement by ships) is low (16% per annum), and that disturbance/displacement of red-throated divers by offshore wind farms is likely to be very much less than disturbance/displacement by ships. This suggests that impacts of displacement from offshore wind farms are unlikely to represent levels of mortality anywhere near to the 16% mortality that occurs due to the combination of many natural factors plus disturbance/displacement by ships. In general, seabirds achieve higher body condition during the non-breeding season than they do while breeding, and the ecology of red-throated divers (foraging over large areas in a highly dispersed manner at low densities at sea) strongly suggests that density-dependent mortality is unlikely during the nonbreeding season. On that basis, it is unlikely that displacement by offshore wind farms would result in an additional mortality exceeding 1% of displaced birds, and any impact is more likely to be close to zero. Assuming that 1% of displaced birds die as a consequence of displacement would appear to be highly precautionary. In addition, strong evidence for density-dependent limitation of breeding numbers of red-throated divers suggests that a small increase in winter mortality would have little or no influence on the size of the red-throated diver population because it is likely to be breeding habitat suitability which sets the carrying capacity.

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Norfolk Vanguard REP1-008: Appendix 3.3 Operational Auk and gannet displacement update and clarification

Norfolk Vanguard Limited Reference: ExA; WQAppx 3.3;10.D1.3: Cited in this document as MacArthur Green 2019c.

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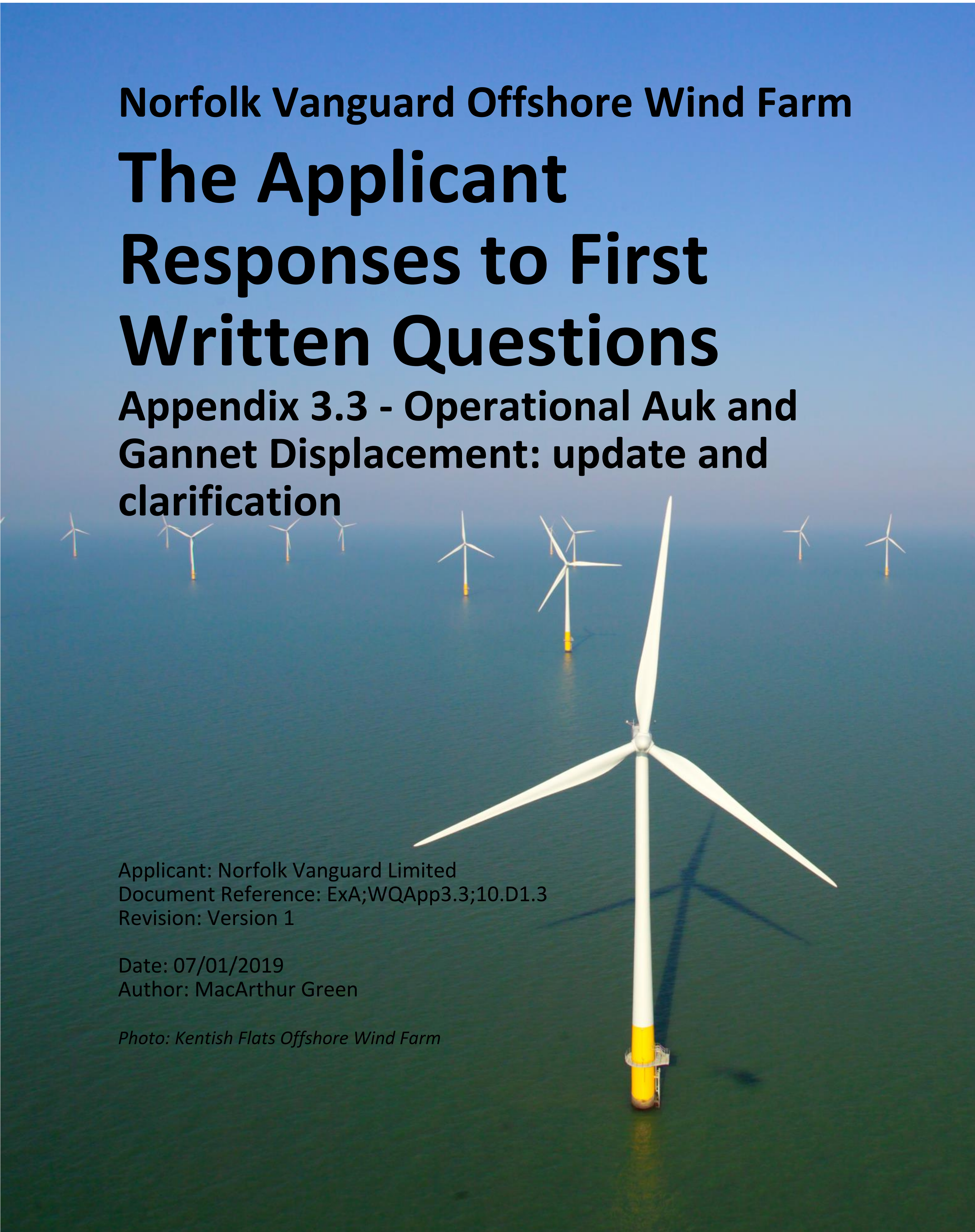

Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.3 - Operational Auk and Gannet Displacement: update and clarification

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.3;10.D1.3
Revision: Version 1

Date: 07/01/2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
12/12/2018	01D	First draft for Norfolk Vanguard Ltd review	MT	JKL, RWF	EV
07/01/2019	02D	Second draft for Norfolk Vanguard Ltd review	MT	RWF	EV
09/01/2019	03	Final version for submission	MT	RWF	EV

Executive Summary

This note provides an update to the operational displacement assessment for Norfolk Vanguard and addresses comments received from Natural England in their Relevant Representation.

A review of evidence for displacement effects for guillemot and razorbill has been undertaken and is included in this note. This review concludes that appropriate (and still precautionary) rates of displacement from wind farms for these species are 50% from within the wind farm itself and 30% within a 1 km buffer, combined with a maximum consequent mortality for displaced individuals of 1%. These rates, as well as the Natural England recommended rates of 30%-70% displacement and 1%-10% mortality have been applied in this update note.

Assessment is also presented using the upper and lower 95% confidence intervals on population abundance for puffin, razorbill, guillemot and gannet for project alone impacts.

Cumulative assessments for puffin, razorbill and guillemot are provided which include the figures presented in the Environmental Statements (ESs) for Hornsea Project 3 and Thanet Extension, and also include figures for the Hywind and Kincardine projects. Natural England also requested a cumulative displacement assessment for gannet, and this will be provided in a subsequent clarification note.

The conclusions of the updated assessment presented in this note remain the same as those in the original Norfolk Vanguard assessment (as presented in the ES), with no impacts greater than minor adverse for any species, either alone or cumulatively.

Table of Contents

Tables

Glossary

BACI	Before-After-Control-Impact
BDMPS	Biologically Defined Minimum Population Scale
CRM	Collision Risk Modelling
ES	Environmental Statement
FFC	Flamborough and Filey Coast (SPA)
HRA	Habitats Regulations Assessment
LSE	Likely Significant Effect
NE	Natural England
NV	Norfolk Vanguard
OWEZ	Offshore Wind Farm Egmond aan Zee
PEIR	Preliminary Environmental Impact Report
RR	Relevant Representation
SNCB	Statutory Nature Conservation Body
SPA	Special Protection Area
TDR	Time-Depth Recorder

1 INTRODUCTION

1. This note provides an update to the Norfolk Vanguard auk (guillemot, razorbill and puffin) and gannet displacement assessment (Vattenfall 2018) and addresses comments from Natural England (NE) in their Relevant Representation for the Norfolk Vanguard application.

2. The detailed comments provided by NE and where they have been addressed are provided in Table 1. Subsequent sections of this note provide updated displacement assessments for each species in relation to the project alone and cumulatively. A review of the evidence for auk displacement and mortality rates is provided in Annex 1.

Table 1 Comments on the auk displacement assessment provided by Natural England (2018) in their relevant representation.

Paragraph	Comment	Response and section of this document where more detail provided
4.2.5	**Assessment of Displacement Impacts** As advised in our Section 42 (Preliminary Environmental Information Report) response, NE require that the variability (uncertainty) in the underlying population estimates (i.e. through consideration of appropriately calculated upper and lower confidence intervals) is considered in the displacement assessments. Whilst the upper and lower confidence limits around the bird abundance estimates are presented in the tables in Annex 1 of Appendix 13.01, these have not been considered in the impact assessments for construction or operational displacement within the ES, with only the mean peak seasonal abundances considered. This approach needs to be revisited for all relevant species.	Additional displacement estimates have been provided in section 1.1 using the lower and upper 95% confidence interval density estimates. In addition, and in-keeping with the use of densities in the collision risk modelling (CRM) displacement using the median density estimates have also been provided. It should be noted that for the auk species considered in this note only the operational assessment has been updated since the construction assessment was accepted by NE.
4.2.14	**Cumulative and In-combination Assessments** We welcome the attempt by the Applicant to include figures for Hornsea 3 and Thanet Extension projects in the cumulative and in-combination assessments of displacement and collision risk. We assume that the figures presented in the assessments for these two sites have been obtained from the PEIRs for these projects, however it would be useful if this could be confirmed by the Applicant. There are a number of methodological issues and uncertainties identified	The cumulative tables provided in this update note (section 1.2) include the figures for Hornsea 3 and Thanet Extension presented in the ESs for those projects. However, it is

Paragraph	Comment	Response and section of this document where more detail provided
	with the baseline data and assessments completed by Hornsea 3 and some methodological issues identified with the assessments for Thanet Extension. Therefore, at this stage the figures for these projects have not been agreed and therefore this will mean that the cumulative and in-combination assessments will require updating during the process once figures for these projects have been agreed. Whilst we acknowledge that this is beyond the Norfolk Vanguard Applicant's control, this means that in addition to the issues noted above, Natural England are currently unable to reach any conclusions on the scale of impact of any cumulative or in-combination displacement and CRM impacts.	acknowledged that these may not be the final figures for these projects. In addition the cumulative tables include the ES estimates for the Hywind and Kincardine projects.
4.2.17	Natural England also suggests that a similar approach to that undertaken for the auk cumulative displacement assessments is undertaken for gannet. This also applies to the assessment of LSE for in-combination assessment of gannet displacement from the FFC SPA.	Assessment of cumulative gannet displacement will be provided in a separate note. Habitats Regulations Assessment (HRA) in relation to the Flamborough and Filey Coast SPA will be provided in a separate note.
4.2.18	The Applicant has considered that a value of 1% mortality when combined with the 70% displacement rate is appropriate for wintering auks. Whilst Natural England agrees that the mortality for auks is likely to be at the low end of the range, we do not agree that using 1% mortality for the cumulative assessment (with 70% displacement) can be considered the worst case scenario. Therefore, our recommendation is a range of mortality rates of 1-10% and displacement rates of 30-70%, with 70% displacement and 10% mortality as the worst case, which is the same as that used by the Applicant in the assessment of auk displacement impacts from the Vanguard project alone.	A review of the evidence for auk displacement from offshore wind farms has been conducted and is provided in Annex 1. This review has found that, on the basis of evidence from existing wind farms, a precautionary displacement rate from the wind farm itself would be 50%, with 30% displaced from a 1 km buffer. The review also concludes that the consequence, in terms of elevated mortality for displaced individuals, is very unlikely to exceed 1% (and that figure remains highly precautionary). Consideration of the impacts of displacement using the
4.2.19	We note that within the Natural England assessment scenario of 30% displacement and 1% mortality to 70% displacement and 10% mortality, a number of the annual predicted addition auk mortalities equates to greater than 1% of baseline mortality of both the largest BDMSP and the biogeographic populations. This is not insignificant and we advise further consideration be given to this once the figures are agreed. This also applies to the assessment of LSE for in-combination assessment of auk displacement from the FFC SPA.	

Paragraph	Comment	Response and section of this document where more detail provided
		range of rates recommended by NE is provided, along with that derived from the evidence review.
Summary of Natural England's key concerns		
3.1	**Lack of consideration of confidence intervals in bird abundance data for displacement assessments** As advised in our Section 42 (PEIR) response, NE require that the variability (uncertainty) in the underlying population estimates (i.e. through consideration of appropriately calculated upper and lower confidence intervals) is considered in the displacement assessments. Whilst the upper and lower confidence limits around the bird abundance estimates are presented in the tables in Annex 1 of Appendix 13.1, these have not been considered in the impact assessments for construction or operational displacement within the Environmental Statement Chapter, with only the mean peak seasonal abundances considered. This approach needs to be revisited for all relevant species. However, as the confidence limits are presented in the tables in Annex 1 of Appendix 13.1, Natural England has undertaken assessments based on these figures as well. We note that for construction displacement, consideration of the range of impacts predicted by considering the confidence limits does not alter the conclusions made by the Applicant for any species for displacement due to construction. The same is true for assessments of operational displacement (with the exception of red-throated diver, but this is largely due to the errors in the abundance data used within the operational matrices and that Natural England does not agree with the displacement and mortality rates used by the Applicant – see below).	Additional displacement estimates have been provided in section 1.1 using the lower and upper 95% confidence interval density estimates. In addition, and in-keeping with the use of densities in the collision risk modelling (CRM) matrices using the median density estimates have been provided. We note that NE has already concluded that this additional precautionary assessment does not change the conclusions for auks (see separate note for consideration of red-throated diver displacement; *Norfolk Vanguard Offshore Wind Farm The Applicant Responses to First Written Questions Appendix 3.1 - Red-throated diver displacement Doc. Ref. ExA;WQApp3.1;10.D1.3*).
5.4	**Auk (puffin, razorbill and guillemot) cumulative and in-combination operational displacement assessments** In addition to the overarching comment above regarding the issues/uncertainties around the data included for Vanguard alone and for Hornsea 3 and Thanet Extension, the Applicant has considered that a value of 1% mortality when combined with the 70% displacement rate is considered appropriate for wintering auks. Natural England notes that definitive mortality rates associated with displacement for seabirds, including auks are not known and	A review of the evidence for auk displacement from offshore wind farms has been conducted and is provided in Annex 1. This review has found that, on the basis of evidence from

Paragraph	Comment	Response and section of this document where more detail provided
	therefore we advise consideration of a range of mortality rates are used in assessments. Whilst Natural England agrees that the mortality for auks is likely to be at the low end of the range, we do not agree that using 1% mortality for the cumulative assessment (with 70% displacement) can be considered the worst case scenario. Therefore, our recommendation is a range of mortality rates of 1-10% and displacement rates of 30-70%, with 70% displacement and 10% mortality as the worst case, which is the same as that used by the Applicant in the assessment of auk displacement impacts from the Vanguard project alone. We note that within the Natural England assessment scenario of 30% displacement and 1% mortality to 70% displacement and 10% mortality, a number of the annual predicted addition auk mortalities equates to greater than 1% of baseline mortality of both the largest BDSMP and the biogeographic populations. This is not insignificant and we advise further consideration be given to this once the figures are agreed. This also applies to the assessment of LSE for in-combination assessment of auk displacement from the FFC SPA. Therefore, we advise that once the figures are agreed and the summed figures accurately presented that the assessment and conclusion of the LSE screening for auk in-combination displacement from FFC SPA is reviewed by the Applicant. We note that the cumulative displacement tables for all three auk species (guillemot, razorbill and puffin) all list the non-breeding seasons for Seagreen Alpha and Bravo as being N/A. We acknowledge that the Environmental Statement (ES) for these projects does not present displacement figures for the non-breeding seasons. However, graphs of monthly abundances of each auk species at each of the project sites across the two survey years are presented in the ES Chapter (Seagreen Wind Energy 2012). These indicate that both guillemot and razorbill were recorded in in all surveys of both Alpha and Bravo during the study period and puffins were recorded in lower numbers in most months. Therefore, consideration should be given to this in the cumulative assessments.	existing wind farms, a precautionary displacement rate from the wind farm itself would be 50%, with 30% from a 1 km buffer. The review also concludes that the consequence, in terms of elevated mortality for displaced individuals, is very unlikely to exceed 1% (and that figure remains highly precautionary). Consideration of the impacts of displacement using the range of rates recommended by NE is provided, along with that derived from the evidence review. HRA in relation to the Flamborough and Filey Coast SPA is provided in a separate note. Further review of the assessments for the Seagreen projects has been undertaken in order to determine appropriate estimates to be used for these projects in the cumulative assessment (section 1.2).

1.1 Operational displacement including uncertainty in density estimates

3. The displacement assessments presented in the ES (Vattenfall 2018) used the peak mean abundance in each biological season within the wind farm and 2 km buffer for all species considered at risk of displacement effects. NE requested that these assessments should also include consideration of uncertainty in the abundance estimates by providing additional displacement predictions obtained using the lower and upper 95% confidence intervals on the abundance estimates.

4. The rates of displacement and mortality recommended by the Statutory Nature Conservation Bodies (SNCBs) for auks are 30-70% displacement and 1-10% mortality (of displaced individuals) and for gannet 60-80% displacement and 1% mortality (NB: red-throated diver is also considered at risk of displacement effects and has been assessed in a separate note: Norfolk Vanguard Offshore Wind Farm The Applicant Responses to First Written Questions Appendix 3.1 - Red-throated diver displacement. Doc. Ref. ExA;WQApp3.1;10.D1.3). Displacement mortality calculated using the lower and upper bounds defined by these rates (30%-1% and 70%-10%) are provided in Table 2, Table 3 and Table 4, together with the abundance estimates used.
5. In addition, a comprehensive review of evidence relating to auk displacement from offshore wind farms (Annex 1), presents evidence for a displacement rate of 50% for birds within the wind farm and 30% within a 1 km buffer, both combined with a highly precautionary maximum mortality of 1%. As estimates of abundance within a 1 km buffer of the wind farm have not been calculated the following assessment update is based on the original 2 km buffer (i.e. it retains precaution due to both the larger buffer and also the application of a single displacement rate across both the wind farm and buffer). The evidence review also reported studies which suggest the potential for habituation to wind farms. This has not been considered in the assessment, however it indicates yet more precaution in the assessment.
6. It is also noted that, despite the figures in Tables 2 to 5 not having been presented in the ES, Natural England stated in their RR that they had taken the uncertainty into account in the auk displacement assessments and this did not change the conclusions presented in the ES. Thus, the project alone operational displacement impacts remain **negligible to minor adverse**.

Table 2. Puffin abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at: 30% - 1%	50%- 1%	70%- 10%
Puffin	East	Breeding	Lwr 95%	0	0	0	0
			Median	40	0.1	0.2	2.8
			Mean	67	0.2	0.3	4.7
			Upr 95%	191	0.6	1	13.4
		Nonbreeding	Lwr 95%	0	0	0	0
			Median	32	0.1	0.2	2.2
			Mean	112	0.3	0.6	7.8
			Upr 95%	417	1.3	2.1	29.2
	West	Breeding	N/A	0	0	0	0
		Nonbreeding	N/A	0	0	0	0
	East & West	Annual	Lwr 95%	0	0	0	0
			Median	72	0.2	0.4	5
			Mean	179	0.5	0.9	12.5
			Upr 95%	608	1.8	3	42.6

Table 3. Razorbill abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at: 30% - 1%	50%- 1%	70%- 10%
Razorbill	East	Breeding	Lwr 95%	156	0.5	0.8	10.9
			Median	526	1.6	2.6	36.8
			Mean	599	1.8	3	41.9
			Upr 95%	1150	3.5	5.8	80.5
		Autumn	Lwr 95%	229	0.7	1.1	16
			Median	520	1.6	2.6	36.4
			Mean	491	1.5	2.5	34.4
			Upr 95%	786	2.4	3.9	55
		Winter	Lwr 95%	74	0.2	0.4	5.2
			Median	291	0.9	1.5	20.4
			Mean	279	0.8	1.4	19.5
			Upr 95%	543	1.6	2.7	38
		Spring	Lwr 95%	212	0.6	1.1	14.8
			Median	762	2.3	3.8	53.3
			Mean	752	2.3	3.8	52.6
			Upr 95%	1302	3.9	6.5	91.1
	West	Breeding	Lwr 95%	96	0.3	0.5	6.7
			Median	259	0.8	1.3	18.1
			Mean	280	0.8	1.4	19.6
			Upr 95%	523	1.6	2.6	36.6
		Autumn	Lwr 95%	89	0.3	0.4	6.2
			Median	345	1	1.7	24.2
			Mean	375	1.1	1.9	26.3
			Upr 95%	729	2.2	3.6	51
		Winter	Lwr 95%	179	0.5	0.9	12.5
			Median	366	1.1	1.8	25.6
			Mean	348	1	1.7	24.4
			Upr 95%	495	1.5	2.5	34.7
		Spring	Lwr 95%	89	0.3	0.4	6.2
			Median	168	0.5	0.8	11.8
			Mean	172	0.5	0.9	12
			Upr 95%	269	0.8	1.3	18.8
	East & West	Annual	Lwr 95%	1124	3.4	5.6	78.7
			Median	3237	9.7	16.2	226.6
			Mean	3296	9.9	16.5	230.7
			Upr 95%	5797	17.4	29	405.8

Table 4. Guillemot abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at: 30% - 1%	50%- 1%	70%- 10%
Guillemot	East	Breeding	Lwr 95%	544	1.6	2.7	38.1
			Median	2853	8.6	14.3	199.7
			Mean	2931	8.8	14.7	205.2
			Upr 95%	5629	16.9	28.1	394
		Nonbreeding	Lwr 95%	1377	4.1	6.9	96.4
			Median	1992	6	10	139.4
			Mean	2197	6.6	11	153.8
			Upr 95%	3441	10.3	17.2	240.9
	West	Breeding	Lwr 95%	439	1.3	2.2	30.7
			Median	1267	3.8	6.3	88.7
			Mean	1389	4.2	6.9	97.2
			Upr 95%	2493	7.5	12.5	174.5
		Nonbreeding	Lwr 95%	1220	3.7	6.1	85.4
			Median	2371	7.1	11.9	166
			Mean	2579	7.7	12.9	180.5
			Upr 95%	4083	12.2	20.4	285.8
	East & West	Annual	Lwr 95%	3580	10.7	17.9	250.6
			Median	8483	25.4	42.4	593.8
			Mean	9096	27.3	45.5	636.7
			Upr 95%	15646	46.9	78.2	1095.2

Table 5. Gannet abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at: 60% - 1%	80% - 1%
Gannet	East	Breeding	Lwr 95%	27	0.2	0.2
			Median	149	0.9	1.2
			Mean	162	1	1.3
			Upr 95%	328	2	2.6
		Autumn	Lwr 95%	816	4.9	6.5
			Median	1375	8.3	11
			Mean	1630	9.8	13
			Upr 95%	2854	17.1	22.8
		Spring	Lwr 95%	0	0	0
			Median	557	3.3	4.5
			Mean	419	2.5	3.4
			Upr 95%	773	4.6	6.2
	West	Breeding	Lwr 95%	9	0.1	0.1
			Median	55	0.3	0.4
			Mean	95	0.6	0.8
			Upr 95%	241	1.4	1.9

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at: 60% - 1%	80% - 1%
		Autumn	Lwr 95%	666	4	5.3
			Median	822	4.9	6.6
			Mean	823	4.9	6.6
			Upr 95%	1013	6.1	8.1
		Spring	Lwr 95%	0	0	0
			Median	9	0.1	0.1
			Mean	18	0.1	0.1
			Upr 95%	65	0.4	0.5
	East & West	Annual	Lwr 95%	1518	9.1	12.1
			Median	2967	17.8	23.7
			Mean	3147	18.9	25.2
			Upr 95%	5274	31.6	42.2

1.2 Cumulative displacement for auks

7. The Norfolk Vanguard cumulative displacement assessment for auks presented effects based on a 70% displacement rate and 1% mortality rate. Natural England requested the assessment should consider more precautionary estimates of displacement and mortality, specifically displacement between 30% and 70% and with mortality between 1% and 10%, affecting all individuals within 2 km of the wind farm boundary.

8. The evidence review provided in Annex 1 found that a precautionary basis for assessment is 50% displacement and a maximum of 1% mortality from within the wind farm and 30% displacement (and a maximum of 1% mortality) within the 1 km buffer of the wind farm. However, the updated displacement matrices provided below present the full ranges of displacement (0-100%) and mortality (0-100%). Thus, displacement impact predictions using both the precautionary rates advised by NE and the evidence-based ones (Annex 1) are presented.

9. Natural England requested updated estimates for several wind farms (Hornsea THREE, Thanet Extension and SeaGreen Alpha and Bravo), although NE also acknowledged that the first two of these were subject to ongoing discussions with the respective developers.

10. The magnitude of additional mortality for each auk species' largest BDMPS which would increase the background mortality by up to 3% is presented in Table 6.

Table 6. Auk populations in UK North Sea waters (see Furness 2015) used in the displacement assessment, the baseline mortality averaged across age classes (Error! Reference source not found.**) and the additional mortality which would increase the baseline rate by 1%, 2% and 3%.**

Species	Largest BDMPS	Average baseline mortality	Magnitude of additional mortality which increases baseline rate by:		
			1%	2%	3%
Guillemot	2,045,078	0.140	2,863	5,726	8,589
Razorbill	591,874	0.174	1,030	2,060	3,090
Puffin	868,689	0.167	1,451	2,901	4,352

1.2.1.1.1 Puffin

11. Norfolk Vanguard East and Norfolk Vanguard West are located beyond the mean maximum foraging range of any puffin breeding colonies. Outside the breeding season, puffins disperse from their breeding sites. Large numbers are found throughout the North Sea in the nonbreeding season (defined as August to February). It was during this period that numbers peaked on the Norfolk Vanguard East site with a mean maximum of 112 individuals (Table 7). The totals at risk on other North Sea wind farms are also presented in Table 7.

Table 7. Cumulative puffin numbers on wind farms in the North Sea.

Project	Breeding season	Non-breeding season
Aberdeen	42.0	81.7
Beatrice	2858.0	2434.8
Blyth Demonstration	235.0	122.8
Dogger Bank Creyke Beck A	37.0	295.2
Dogger Bank Creyke Beck B	102.0	742.9
Dogger Bank Teesside A	34.0	273.0
Dogger Bank Teesside B	35.0	328.7
Dudgeon	1.0	3.2
East Anglia ONE	16.0	32.0
East Anglia THREE	181.0	307.0
Galloper	0.0	0.8
Greater Gabbard	0.0	0.9
Hornsea Project One	1070.0	1257.0
Hornsea Project Two	468.0	2039.0
Hornsea Project Three	253.0	127.0
Humber Gateway	15.0	9.6
Hywind	119.0	85.0
Inch Cape	2956.0	2688.0
Kincardine	19.0	0
Lincs and LID6	3.0	6.0
London Array I & II	0.0	0.6
Moray East	2795.0	656.4

Project	Breeding season	Non-breeding season
Neart na Gaoithe	2562.0	2103.4
Race Bank	1.0	9.6
Seagreen A	2572.0	1526.0
Seagreen B	3582.0	3863.0
Sheringham Shoal	4.0	25.8
Teesside	35.0	18.0
Thanet	0.0	0.1
Thanet Extension	0.0	0.0
Triton Knoll*	23.0	70.7
Westermost Rough	61.0	35.0
Seasonal Total (Ex. NV)	**20079.0**	**19143.2.8**
Annual Total (Ex. NV)		**39222**
Norfolk Vanguard East	0	112
Norfolk Vanguard West	0	0
Seasonal Total (Inc. NV)	**20079**	**19255**
Annual Total (Inc. NV)		**39334**

12. Natural England does not consider a single combination of displacement and mortality in their assessment of impact, instead advising presentation of the ranges from 0 to 100% as provided in this note (e.g. Table 8), with a focus on 30% to 70% displacement and 1%-10% mortality. However, evidence in support of the use of a precautionary displacement rate of 50% within the wind farm, 30% within the 1km buffer and 0% thereafter, combined with a 1% mortality rate for guillemot and razorbill (Annex 1) is also considered appropriate for puffin. For the current assessment presented here application of this level of impact indicates that the baseline mortality rate for the relevant populations (North Sea BDMPS) would increase by less than 1% (Table 8).

13. Consequently, the potential cumulative annual displacement mortality for puffin would not materially alter the background mortality of the population and would be undetectable. Therefore, the magnitude of effect is assessed as negligible. As the species is of low to medium sensitivity to disturbance, the impact significance is **negligible to minor adverse**.

Table 8. Puffin cumulative displacement matrix. Levels of mortality which would increase the baseline mortality by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%:

		Mortality (%)																			
		1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	100
Displacement (%)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2	8	16	24	31	39	47	55	63	71	79	157	197	236	315	393	472	551	629	708	787
	4	16	31	47	63	79	94	110	126	142	157	315	393	472	629	787	944	1101	1259	1416	1573
	6	24	47	71	94	118	142	165	189	212	236	472	590	708	944	1180	1416	1652	1888	2124	2360
	8	31	63	94	126	157	189	220	252	283	315	629	787	944	1259	1573	1888	2203	2517	2832	3147
	10	39	79	118	157	197	236	275	315	354	393	787	983	1180	1573	1967	2360	2753	3147	3540	3933
	12	47	94	142	189	236	283	330	378	425	472	944	1180	1416	1888	2360	2832	3304	3776	4248	4720
	14	55	110	165	220	275	330	385	441	496	551	1101	1377	1652	2203	2753	3304	3855	4405	4956	5507
	16	63	126	189	252	315	378	441	503	566	629	1259	1573	1888	2517	3147	3776	4405	5035	5664	6293
	18	71	142	212	283	354	425	496	566	637	708	1416	1770	2124	2832	3540	4248	4956	5664	6372	7080
	20	79	157	236	315	393	472	551	629	708	787	1573	1967	2360	3147	3933	4720	5507	6293	7080	7867
	25	98	197	295	393	492	590	688	787	885	983	1967	2458	2950	3933	4917	5900	6883	7867	8850	9833
	30	118	236	354	472	590	708	826	944	1062	1180	2360	2950	3540	4720	5900	7080	8260	9440	10620	11800
	40	157	315	472	629	787	944	1101	1259	1416	1573	3147	3933	4720	6293	7867	9440	11013	12587	14160	15734
	50	197	393	590	787	983	1180	1377	1573	1770	1967	3933	4917	5900	7867	9833	11800	13767	15734	17700	19667
	60	236	472	708	944	1180	1416	1652	1888	2124	2360	4720	5900	7080	9440	11800	14160	16520	18880	21240	23600
	70	275	551	826	1101	1377	1652	1927	2203	2478	2753	5507	6883	8260	11013	13767	16520	19274	22027	24780	27534
	80	315	629	944	1259	1573	1888	2203	2517	2832	3147	6293	7867	9440	12587	15734	18880	22027	25174	28320	31467
	90	354	708	1062	1416	1770	2124	2478	2832	3186	3540	7080	8850	10620	14160	17700	21240	24780	28320	31860	35400
	100	393	787	1180	1573	1967	2360	2753	3147	3540	3933	7867	9833	11800	15734	19667	23600	27534	31467	35400	39334

1.2.1.1.2 Razorbill

14. Norfolk Vanguard East and Norfolk Vanguard West are located beyond the mean maximum foraging range of any razorbill breeding colonies. Outside the breeding season, razorbills migrate from their breeding sites. Large numbers are found throughout the North Sea in the nonbreeding seasons (covering the period from August to March). The annual total of razorbills at risk of displacement on the Norfolk Vanguard site (combined across the breeding season and all the nonbreeding seasons) was a mean maximum of 3,296 individuals (Table 9). The totals at risk on other North Sea wind farms are also presented in Table 9.

Table 9. Cumulative razorbill numbers on wind farms in the North Sea.

Project	Breeding season	Post-breeding season	Non-breeding season	Pre-breeding season
Aberdeen	161.0	64.4	7.3	25.7
Beatrice	873.0	833.0	555.3	833.0
Blyth Demonstration	121.0	90.9	60.6	90.9
Dogger Bank Creyke Beck A	1250.0	1576.0	1728.0	4149.0
Dogger Bank Creyke Beck B	1538.0	2097.0	2143.0	5118.7
Dogger Bank Teesside A	834.0	310.3	958.5	1919.0
Dogger Bank Teesside B	1153.0	592.3	1426.0	2953.3
Dudgeon	256.0	346.1	745.4	346.1
East Anglia ONE	16.0	26.0	154.5	336.0
East Anglia THREE	1807.0	1122.0	1499.0	1524.0
Galloper	44.0	43.0	105.5	394.0
Greater Gabbard	0.0	0.0	387.3	83.8
Hornsea Project One	1109.0	4812.3	1517.5	1802.8
Hornsea Project Two	2511.0	4220.5	719.5	1668.0
Hornsea Project Three	630.0	2020.0	3649.0	1236.0
Humber Gateway	27.0	20.0	13.4	20.0
Hywind	30	719.0	10	0
Inch Cape	1436.0	2870.0	651.0	N/A
Kincardine	2.0	0	0	0
Lincs and LID6	45.0	33.5	22.3	33.5
London Array I & II	14.0	20.4	13.6	20.4
Moray	2423.0	1102.6	30.2	168.3
Neart na Gaoithe	331.0	5492.4	507.8	
Race Bank	28.0	42.0	28.0	42.0
Seagreen A	5876		1003	
Seagreen B	3698		1272	
Sheringham Shoal	106.0	1343.0	211.3	30.2
Teesside	16.0	61.5	1.9	20.0
Thanet	3.0	0.0	13.6	20.9
Thanet Extension	N/A	N/A	34.0	50.0
Triton Knoll*	40.0	253.7	854.5	116.7
Westermost Rough	91.0	121.3	151.6	90.9
Seasonal Total (Ex. NV)	**26469**	**30233.2**	**20474.6**	**23093**

Project	Breeding season	Post-breeding season	Non-breeding season	Pre-breeding season
Annual Total (Ex. NV)				**100270**
Norfolk Vanguard East	**599**	**491**	**279**	**752**
Norfolk Vanguard West	**280**	**375**	**348**	**172**
Seasonal Total (Inc. NV)	**27368**	**31099.2**	**21101.6**	**24017**
Annual Total (Inc. NV)				**103586**

15. Natural England does not consider a single combination of displacement and mortality in their assessment of impact, instead advising presentation of the ranges from 0 to 100% as provided in this note, with a focus on 30% to 70% displacement and 1%-10% mortality. However, evidence is provided in Annex 1 in support of the use of a precautionary displacement rate of 50% with a 1% mortality rate for guillemot and razorbill. For the current assessment presented here application of this level of impact indicates that the baseline mortality rate for the relevant populations (North Sea BDMPS) would increase by less than 1% (Table 10).

16. Consequently, the potential cumulative annual displacement mortality for razorbill would not materially alter the background mortality of the population and would be undetectable. Therefore, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

Table 10. Razorbill cumulative displacement matrix. Levels of mortality which would increase the baseline mortality by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%:

Displacement (%)	Mortality (%)																			
	1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	100
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	21	41	62	83	104	124	145	166	186	207	414	518	622	829	1036	1243	1450	1657	1865	2072
4	41	83	124	166	207	249	290	331	373	414	829	1036	1243	1657	2072	2486	2900	3315	3729	4143
6	62	124	186	249	311	373	435	497	559	622	1243	1554	1865	2486	3108	3729	4351	4972	5594	6215
8	83	166	249	331	414	497	580	663	746	829	1657	2072	2486	3315	4143	4972	5801	6629	7458	8287
10	104	207	311	414	518	622	725	829	932	1036	2072	2590	3108	4143	5179	6215	7251	8287	9323	10359
12	124	249	373	497	622	746	870	994	1119	1243	2486	3108	3729	4972	6215	7458	8701	9944	11187	12430
14	145	290	435	580	725	870	1015	1160	1305	1450	2900	3626	4351	5801	7251	8701	10151	11602	13052	14502
16	166	331	497	663	829	994	1160	1326	1492	1657	3315	4143	4972	6629	8287	9944	11602	13259	14916	16574
18	186	373	559	746	932	1119	1305	1492	1678	1865	3729	4661	5594	7458	9323	11187	13052	14916	16781	18645
20	207	414	622	829	1036	1243	1450	1657	1865	2072	4143	5179	6215	8287	10359	12430	14502	16574	18645	20717
25	259	518	777	1036	1295	1554	1813	2072	2331	2590	5179	6474	7769	10359	12948	15538	18128	20717	23307	25896
30	311	622	932	1243	1554	1865	2175	2486	2797	3108	6215	7769	9323	12430	15538	18645	21753	24861	27968	31076
40	414	829	1243	1657	2072	2486	2900	3315	3729	4143	8287	10359	12430	16574	20717	24861	29004	33147	37291	41434
50	518	1036	1554	2072	2590	3108	3626	4143	4661	5179	10359	12948	15538	20717	25896	31076	36255	41434	46614	51793
60	622	1243	1865	2486	3108	3729	4351	4972	5594	6215	12430	15538	18645	24861	31076	37291	43506	49721	55936	62152
70	725	1450	2175	2900	3626	4351	5076	5801	6526	7251	14502	18128	21753	29004	36255	43506	50757	58008	65259	72510
80	829	1657	2486	3315	4143	4972	5801	6629	7458	8287	16574	20717	24861	33147	41434	49721	58008	66295	74582	82869
90	932	1865	2797	3729	4661	5594	6526	7458	8390	9323	18645	23307	27968	37291	46614	55936	65259	74582	83905	93227
100	1036	2072	3108	4143	5179	6215	7251	8287	9323	10359	20717	25896	31076	41434	51793	62152	72510	82869	93227	103586

1.2.1.1.3 Guillemot

17. Norfolk Vanguard East and Norfolk Vanguard West are located beyond the mean maximum foraging range of any guillemot breeding colonies. Outside the breeding season, guillemots disperse from their breeding sites. Large numbers are found throughout the North Sea in the nonbreeding season (defined as August to February). It was during this period that numbers peaked on the Norfolk Vanguard site with a mean maximum of 4,776 individuals (Table 11). The totals at risk on other North Sea wind farms are also presented in Table 11.

Table 11. Cumulative guillemot numbers on North Sea wind farms.

Project	Breeding season	Non-breeding season
Aberdeen	547.0	225.0
Beatrice	13610.0	2755.0
Blyth Demonstration	1220.0	1321.0
Dogger Bank Creyke Beck A	5407.0	6142.0
Dogger Bank Creyke Beck B	9479.0	10621.0
Dogger Bank Teesside A	3283.0	2268.0
Dogger Bank Teesside B	5211.0	3701.0
Dudgeon	334.0	542.0
East Anglia ONE	274.0	640.0
East Anglia THREE	1744.0	2859.0
Galloper	305.0	593.0
Greater Gabbard	345.0	548.0
Hornsea Project One	9836.0	8097.0
Hornsea Project Two	7735.0	13164.0
Hornsea Project Three	13374.0	17772.0
Humber Gateway	99.0	138.0
Hywind	249	2136
Inch Cape	4371.0	3177.0
Kincardine	632	0
Lincs and LID6	582.0	814.0
London Array I & II	192.0	377.0
Moray	9820.0	547.0
Neart na Gaoithe	1755.0	3761.0
Race Bank	361.0	708.0
Seagreen A	13606	4688
Seagreen B	11118.0	4112
Sheringham Shoal	390.0	715.0
Teesside	267.0	901.0
Thanet	18.0	124.0
Thanet Extension	49.0	837.0
Triton Knoll*	425.0	746.0
Westermost Rough	347.0	486.0
Seasonal Total (Ex. NV)	116985	95515
Annual Total (Ex. NV)		**208513**
Norfolk Vanguard East	**2931**	**2197**

Project	Breeding season	Non-breeding season
Norfolk Vanguard West	**1389**	**2579**
Seasonal Total (Inc. NV)	**121305**	**100291**
Annual Total (Inc. NV)		**221596**

18. Natural England does not consider a single combination of displacement and mortality in their assessment of impact, instead advising presentation of the ranges from 0 to 100% as provided in this note, with a focus on 30% to 70% displacement and 1%-10% mortality. However, evidence is provided in Annex 1 in support of the use of a precautionary displacement rate of 50% with a 1% mortality rate for guillemot and razorbill. For the current assessment presented here application of this level of impact indicates that the baseline mortality rate for the relevant populations (North Sea BDMPS) would increase by less than 1% (Table 12).

19. Consequently, the potential cumulative annual displacement mortality for razorbill would not materially alter the background mortality of the population and would be undetectable. Therefore, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

Table 12 Guillemot cumulative displacement matrix. Levels of mortality which would increase the baseline mortality by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%:

Displacement (%)	Mortality (%)																			
	1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	100
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	44	89	133	177	222	266	310	355	399	443	886	1108	1330	1773	2216	2659	3102	3546	3989	4432
4	89	177	266	355	443	532	620	709	798	886	1773	2216	2659	3546	4432	5318	6205	7091	7977	8864
6	133	266	399	532	665	798	931	1064	1197	1330	2659	3324	3989	5318	6648	7977	9307	10637	11966	13296
8	177	355	532	709	886	1064	1241	1418	1595	1773	3546	4432	5318	7091	8864	10637	12409	14182	15955	17728
10	222	443	665	886	1108	1330	1551	1773	1994	2216	4432	5540	6648	8864	11080	13296	15512	17728	19944	22160
12	266	532	798	1064	1330	1595	1861	2127	2393	2659	5318	6648	7977	10637	13296	15955	18614	21273	23932	26592
14	310	620	931	1241	1551	1861	2172	2482	2792	3102	6205	7756	9307	12409	15512	18614	21716	24819	27921	31023
16	355	709	1064	1418	1773	2127	2482	2836	3191	3546	7091	8864	10637	14182	17728	21273	24819	28364	31910	35455
18	399	798	1197	1595	1994	2393	2792	3191	3590	3989	7977	9972	11966	15955	19944	23932	27921	31910	35899	39887
20	443	886	1330	1773	2216	2659	3102	3546	3989	4432	8864	11080	13296	17728	22160	26592	31023	35455	39887	44319
25	554	1108	1662	2216	2770	3324	3878	4432	4986	5540	11080	13850	16620	22160	27700	33239	38779	44319	49859	55399
30	665	1330	1994	2659	3324	3989	4654	5318	5983	6648	13296	16620	19944	26592	33239	39887	46535	53183	59831	66479
40	886	1773	2659	3546	4432	5318	6205	7091	7977	8864	17728	22160	26592	35455	44319	53183	62047	70911	79775	88638
50	1108	2216	3324	4432	5540	6648	7756	8864	9972	11080	22160	27700	33239	44319	55399	66479	77559	88638	99718	110798
60	1330	2659	3989	5318	6648	7977	9307	10637	11966	13296	26592	33239	39887	53183	66479	79775	93070	106366	119662	132958
70	1551	3102	4654	6205	7756	9307	10858	12409	13961	15512	31023	38779	46535	62047	77559	93070	108582	124094	139605	155117
80	1773	3546	5318	7091	8864	10637	12409	14182	15955	17728	35455	44319	53183	70911	88638	106366	124094	141821	159549	177277
90	1994	3989	5983	7977	9972	11966	13961	15955	17949	19944	39887	49859	59831	79775	99718	119662	139605	159549	179493	199436
100	2216	4432	6648	8864	11080	13296	15512	17728	19944	22160	44319	55399	66479	88638	110798	132958	155117	177277	199436	221596

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Annex 1

Are guillemots and razorbills displaced from operating offshore wind farms?

1. Garthe and Hüppop (2004), and Furness et al. (2013), identified guillemots and razorbills as among the seabird species most likely to be displaced by offshore wind farms. However, those assessments were based on knowledge of seabird ecology, and without evidence from operational offshore wind farms. Dierschke et al. (2016) reviewed evidence for displacement obtained from studies published up to 2016 that compared seabird abundances within and outside European offshore wind farms post-construction with baseline data from before construction. Studies at twenty different operational offshore wind farms found guillemots and/or razorbills present at twelve of these in large enough numbers for robust statistical analysis of avoidance or attraction, while in a further four cases the evidence was poor and inconclusive.

2. Guillemots were strongly displaced from five, weakly displaced from two, unaffected at two, strongly attracted at one, and at two others there was weak evidence of slight attraction. Razorbills were strongly displaced from two, weakly displaced from three, unaffected at two, not significantly attracted at any, but there was weak evidence of slight attraction at one. Large auks (guillemots or razorbills combined) were weakly displaced at two sites additional to those included for the single-species results, and there was weak evidence of slight displacement at another site.

3. The evidence is clear; guillemots and razorbills do tend to avoid offshore wind farms, but avoidance is incomplete and highly variable among sites. This very variable behavioural response may differ depending on ecological conditions. For example, birds may be displaced more strongly at times of year when they are not constrained, whereas during periods of stress they may be unwilling to be displaced. Variability in displacement may also be a response to configuration of wind farm sites. For example, Leopold et al. (2013) suggested that the lower displacement of alcids from Egmond aan Zee (OWEZ) than from the nearby Princess Amalia was likely to be due to closer spacing of turbines at Princess Amalia than at OWEZ, allowing more alcids to utilize the site with less dense turbine spacing.

How strong is the displacement effect?

4. Overall, Dierschke et al. (2016) concluded that the mean outcome across all offshore wind farms was ‘weak displacement’ for guillemot and for razorbill. They defined this, where the change in density was statistically significant, as less than an average of 50% reduction in density post-construction compared to pre-construction data.

5. The strongest displacement effect was reported for Thorntonbank and Bligh Bank offshore wind farms in Belgian waters (Dierschke et al. 2016). This study, used a Before-After-Control-Impact (BACI) design with monthly boat-based surveys using

Distance correction and spanning several years of data collection post-construction (Vanermen et al. 2016). There was a significant reduction, of 68%, in guillemot density but no significant reduction in razorbill density (but a non-significant reduction of 55%) post-construction at Thorntonbank (including a buffer zone of 0.5 km outside the wind farm). There was no significant reduction in guillemot or razorbill density in a buffer zone from 0.5 km to 3 km from the outer turbines (but they report a non-significant reduction of 24% in guillemot density and a non-significant increase of <10% in razorbill density in the outer buffer zone).

6. At Bligh Bank, there was a significant reduction, of 75%, in guillemot density and a significant reduction, of 67%, in razorbill density (including a buffer zone of 0.5 km outside the wind farm). In a buffer zone from 0.5 km to 3 km from the outer turbines at Bligh Bank there was a smaller (49%), but statistically significant, reduction in guillemot density, but no significant difference in razorbill density (but a non-significant reduction of 32%) between pre-construction and post-construction. Vanermen et al. (2014) reported displacement of guillemots at Bligh Bank as 71%, and showed that razorbill numbers at Bligh Bank increased between pre-construction and post-construction. This is largely the same data set as reported above in the analysis by Vanermen et al. (2016), so it is interesting to note the different conclusion regarding razorbill, which suggests variation across years in relation to ecological conditions.

7. From data collected during boat-based surveys over three years at alpha ventus in the German North Sea, Welcker and Nehls (2016) found that alcids (guillemots and razorbills combined) showed a significant avoidance of the wind farm. They reported a reduction in density of about 75% inside the wind farm plus a buffer of 300 m, compared with densities more than 2.5 km from the wind farm. The data indicated progressively less avoidance with increasing distance up to 2.5 km, after which no effect was apparent. The decrease in density from zero to 2.5 km from the wind farm was about 30% on average, so considerably less than seen within the wind farm.

8. From data collected by boat-based surveys, with Distance correction, in a large area during pre-construction and post-construction periods for two offshore wind farms, Leopold et al. (2013) reported statistically significant reduction in density of guillemots at Princess Amalia and OWEZ. Razorbill densities were significantly reduced at Princess Amalia, but there was no significant reduction at OWEZ, and the non-significant trend was close to zero for that site. Lindeboom et al. (2011) concluded that for these sites the counts in and around the wind farm indicated no marked avoidance by guillemots or razorbills. The percentage displacement of guillemots was not quantified by Leopold et al. (2013), but appeared to be between 30% and 70% at the wind farms. Percentage displacement of razorbills was not

quantified but appeared to be between 30% and 70% at Princess Amalia, but between 0% and 20% at OWEZ. There was no evidence of significant displacement from the buffer zone surrounding these two wind farms. Both species showed similar, or slightly stronger, displacement from an area used as a ship park within the survey area, indicating that alcids may show at least as strong displacement from other large structures in the marine environment as from offshore wind turbines (Leopold et al. 2013).

9. Fox et al. (2006) and Petersen et al. (2006) state that there was no statistically significant change in guillemot density within Horns Rev 1 wind farm post-construction compared to pre-construction. However, they did report that flying guillemots tended to show macro-avoidance, some birds flying around the wind farm rather than through it. There was a small and non-significant decrease in relative abundance of guillemots within the wind farm area, suggesting a possible small and incomplete displacement effect.

10. No displacement of guillemots was found as a consequence of construction of Thanet offshore wind farm, while displacement of razorbills was statistically significant but at a level of less than 50% (Ecology Consulting 2012, Percival 2013).

11. APEM (2016) showed that guillemot density decreased in the vicinity of London Array offshore wind farm during construction, but there were not sufficient post-construction data to reach a clear conclusion as to the extent of avoidance post-construction, although post-construction distribution appeared to indicate partial displacement.

12. At North Hoyle offshore wind farm, guillemot numbers within the wind farm increased by 55% post-construction compared to pre-construction (PMSS 2007).

13. At other sites, displacement of guillemots and/or razorbills was either less than 50%, or was highly variable so difficult to detect statistically and difficult to quantify, as summarised for each site by Dierschke et al. (2016).

14. Since that review, Vallejo et al. (2017) have reported on studies of guillemot numbers across an area of 360 km^2 that included Robin Rigg offshore wind farm. They concluded that relative abundance of guillemots remained similar within the Robin Rigg offshore wind farm across all development phases, and that their data show no significant displacement of guillemots by that wind farm. Indeed, guillemot relative density within the wind farm was marginally, but not significantly, higher post-construction than pre-construction.

15. It is clear that guillemots and razorbills are incompletely displaced from some offshore wind farms, and apparently not displaced at all from some. On average,

displacement results in densities within offshore wind farms that are about 50% of the density in the wider area around these sites. In some cases there is some slight displacement of birds from a buffer zone surrounding the outer turbines, but this also seems to vary among sites. The buffer zone where densities are reduced is generally less than 2 km wide, and in most cases appears to be no more than 500 m wide. Where it has been measured, guillemot and razorbill density increases across the buffer zone, with distance from the turbines, up to the 'background' density. Displacement results in densities within 2 km of the offshore wind farms being reduced by less than 30%, and most studies report no significant reduction in density within the buffer zone.

16. An evidence-based, but still precautionary, assessment of displacement of alcids by offshore wind farms might assume that alcid densities would be reduced inside offshore wind farms by 50% relative to densities in the surrounding area, and by 30%, on average, across a 1 km buffer zone surrounding the wind farm. There are very few examples where displacement is greater than this, and many cases where it is much less.

Is there evidence for habituation of guillemots and razorbills to offshore wind farms?

17. If guillemots and razorbills habituate over time to the presence of offshore wind farms, then habitat loss might be negligible in the long term. In relation to Thorntonbank and Bligh Bank offshore wind farms, Vanermen et al. (2012) concluded 'During recent surveys in 2012, good numbers of auks were encountered inside the wind farm. From an ecological point of view, the presence of auks is very interesting, and we wonder if these self-fishing species are already habituating to the presence of the turbines, and if they will profit from a (hypothetical) increase in food availability'. It has already been seen that cormorants and shags have habituated to offshore wind farms and have learned to aggregate at these sites where they can roost on turbine railings and surfaces, and forage on the aggregations of fish that occur around turbine foundations and scour protection (Dierschke et al. 2016). However, the generally modest increases in fish abundance within wind farms that have been detected so far seem to be mainly for benthic fish such as gadoids and blennies (Bergstrom et al. 2013, van Hal et al. 2017, Stenberg et al. 2015). It remains to be seen whether abundance of pelagic-feeding fish such as sandeels, sprats and young herring that are preferred prey of guillemots and razorbills also regularly show an increased abundance or local concentration within offshore wind farms.

18. Leopold and Verdaat (2018) found some evidence for auks habituating to Luchterduinen offshore wind farm and suggested a methodology to assess the extent of such habituation. They reported that '*on relatively many occasions, birds of*

both these species [guillemots and razorbills] were seen to dive within the wind farm (as well as just outside). We could see no difference in behaviour of these birds inside and outside the wind farm. Diving birds often dived several times in succession, indicating that diving was not a panic reaction in response to them suddenly seeing a working turbine…. This has important implications for how we must judge wind farm effects on seabird ecology. If birds refrain from foraging within wind farms, the entire footprint of any wind farm is lost as feeding habitat. If, on the other hand, birds that are found within the wind farm forage normally, the amount of habitat loss would merely be the footprint of the wind farm times the reduced density of the species involved…. It is also possible that seabirds could learn to (or are already learning to) exploit the new habitat of offshore wind farms to their advantage…. Our observations suggest that they may be on this track: both guillemots and razorbills are now feeding in offshore wind farms'.

19. Evidence for guillemots and razorbills habituating to the presence of operational offshore wind turbines is very limited, but there are hints, as cited above, that this may be occurring. Further evidence of alcid behaviour at operational offshore wind farms would be desirable, potentially following the protocols proposed by Leopold and Verdaat (2018).

What are the likely consequences of displacement for individuals?

20. Displacement could influence individual guillemots or razorbills if offshore wind farm barrier effects or habitat loss result in a change in the bird's energy budget. Under some circumstances, though not all, displacement could increase energy costs, or could result in decreased energy intake. The former could arise if birds had to fly more to avoid offshore wind farms or to reach more distant foraging areas. The latter could arise if displacement was to an area of lower quality habitat where food capture rates were lower, or if displacement resulted in an increase in guillemot or razorbill density on the sea, with a consequent increase in intra-specific competition. Alternatively, displacement may have no effect on individuals if birds are displaced into equally good habitat so that their energy budget is unaffected, or if birds could buffer any impact on energy budget by adjusting their time budget (for example by spending a higher proportion of the time foraging rather than resting, in order to compensate for an increase in energy budget).

21. Density-dependent competition for food could apply in this species during the nonbreeding season if densities of guillemots and razorbills were high enough to deplete their food resource (small fish such as sprats, young herring, sandeels) or to result in reduced catchability of fish due to disturbance competition between foraging alcids. Then if birds had to increase effort to obtain their food requirements, they could potentially reach a limit where they had no more time available to

increase foraging effort further, or were at an energy ceiling that would not permit them to work harder without incurring a loss in body condition (Drent and Daan 1980).

22. Little is known about nonbreeding season energy budgets of guillemots and razorbills, but this information is key to understanding the possible consequences of displacement. If individuals are in relatively good condition during the nonbreeding season and spend only a small proportion of their daily activity budget in foraging, they may have the capacity to buffer against any additional energetic expense of displacement and barrier effects. On the other hand, if individuals have to work to capacity (either in terms of time allocated to foraging or in terms of physiological limits to energy expenditure), then there would be no scope for buffering any additional costs resulting from displacement or barrier effects.

23. Searle et al. (2017) developed an individual-based model to assess impact of displacement and barrier effects for breeding guillemots and razorbills (and puffins and kittiwakes) in the Forth-Tay region during the chick-rearing period. No equivalent model has yet been developed for guillemots and razorbills in the nonbreeding season, due at least in part to the lack of necessary data for the nonbreeding season. Searle's model was tested against scenarios with offshore wind farms in the Forth-Tay region close to the breeding colonies. In the tested scenarios, birds were commuting several times a day from the colony to foraging areas, so potentially affected by barrier effects and displacement; the models assumed a 60% displacement from wind farms plus a buffer zone of 500 m around each wind farm, and a 100% barrier effect (all birds flying around rather than through wind farms).

24. For breeding guillemots, scenarios with offshore wind farms placed relatively close to the study colony (Isle of May) resulted in additional adult guillemot mortality of between 0.003% to 0.31%, depending on modelled food abundance and distribution. For breeding razorbills, scenarios with offshore wind farms placed relatively close to the study colony (Isle of May) resulted in additional adult razorbill mortality of between 0.08% and 0.17%, depending on modelled food abundance and distribution. These scenarios do not apply to birds in the nonbreeding season, but strongly suggest that impacts of displacement and barrier effects for guillemots and razorbills have a very small impact on adult survival, even when tested in scenarios with multiple offshore wind farms placed close to colonies between nest sites and foraging grounds.

25. In the context of overwinter survival, it is relevant that in many seabird species, including alcids, most mortality occurs during winter (e.g. Coulson et al. 1983, Reynolds et al. 2011). This may be caused by a variety of factors, such as winter storms (Anker-Nilssen et al. 2018). However, there is also evidence that seabirds

tend to be heavier in winter than during the breeding season (e.g. Coulson et al. 1983). It is inferred from this that most seabirds have relatively little difficulty in finding enough food during the nonbreeding season so can achieve higher body condition that buffers against short periods of adverse weather conditions. For example, puffins are 20-30% heavier in winter than in summer as a result of storing fat during the nonbreeding season, and the same is true of guillemots (Harris et al. 2000, Anker-Nilssen et al. 2018). An implication is that their body condition may not be greatly affected by plausible levels of displacement or disturbance, since these birds are capable of maintaining high body weight through winter.

26. This leads to consideration of when during the nonbreeding season there may be critical periods that influence survival (bottlenecks). One possibility for alcids is during autumn when birds moult and become temporarily flightless. At this time it is critical that birds are in an area where there is a reliable food supply. There have been ‘wrecks’ of guillemots and razorbills in autumn that have sometimes been suggested as having been related to flightless auks being unable to find food. Many autumn ‘wrecks’ seem to involve mainly juvenile birds, suggesting that the mass mortality of guillemots or razorbills in autumn affects large cohorts of inexperienced young birds that fail to find suitable feeding areas early in life. However, in some cases autumn ‘wrecks’ have involved mostly adult birds, in heavy flight feather moult, suggesting that a spatial mis-match between distributing of moulting adults and a reliable food supply can cause mortality events.

27. Another possible ‘bottleneck’ occurs during mid to late winter, when prey depletion through the nonbreeding season may have occurred and daylength is short and storms may make feeding difficult. Guillemots and razorbills are able to feed at night (for example, guillemots overwinter in the Arctic in places where there is continuous dark for many weeks through mid-winter, and they are able to find food and maintain high body weights during that period). However, it is thought that guillemots and razorbills mainly feed during daylight hours and tend to rest during the night. We are aware of only one study that has collected data on foraging activity of guillemots through the winter by deploying time-depth recorders (TDRs) (Daunt et al. 2007).

28. Daunt et al. (2007) deployed TDRs and geolocators on breeding adult guillemots at the Isle of May in July 2005. Data were recovered from 13 of these birds in June 2006. They had moved into the central North Sea in late summer and autumn, during which period foraging effort (in terms of time spent diving) was low. The birds spent winter (November to March) in the southern North Sea. Foraging effort was slightly higher in November-March than it had been in autumn, though lower in January-February than in November-December. Birds returned to the vicinity of the

Isle of May from late February. Foraging effort peaked in March, but in April was the lowest of the year. These data are from just one nonbreeding period (2005-06), so it is impossible to assess how representative the data are for other years. The data suggest that if there is a 'bottleneck' when foraging is most challenging, that may occur during winter, or possibly in early spring when birds first return to the vicinity of the colony. However, the mean number of dives per 24 hrs varied only between a minimum of 160 during April and a maximum of 280 during March. Diving was rarely recorded at night during April to November, but during December to March dives at night represented about 30% of the total. This also suggests that birds may have been working harder in December to March.

29. Mean foraging depth, and dive duration, showed little seasonal variation, averaging around 20 m and 70 s respectively, but nocturnal dives tended to be much shallower and shorter than daytime dives. Overall, birds spent about 6 to 7 hours in foraging each day, and this showed little seasonal variation apart from being lower in autumn. These figures suggest that there may be considerable flexibility in guillemot time budgets that would potentially allow birds to increase foraging effort if required. It would be very helpful to have data on these nonbreeding season foraging metrics for guillemots and razorbills for a number of years in order to assess whether foraging effort reaches an upper limit in some years, perhaps relating to abundance of food fish stocks or to other environmental factors. But these preliminary data suggest some flexibility and headroom in foraging effort during the nonbreeding period. That tentative conclusion matches general understanding of bird time budgets; larger birds generally spend a smaller percentage of their time in foraging than do smaller birds. This is the case at all times of year, and was demonstrated a long time ago for breeding seabirds.

30. Studying seabirds breeding at the Farne Islands, Northumberland, Pearson (1968) showed that breeding guillemots spent 16% of daylight hours foraging, compared to 4-8% by shags, 37% by puffins, 43-57% by kittiwakes and 54-103% by Arctic terns. This also suggests that, compared to small birds, larger birds, such as guillemots and razorbills, are likely to have more 'spare' time that could be put into foraging if necessary. Studies of breeding guillemots in Shetland further demonstrate the potential for flexibility of foraging effort in this species; in 1991, when sandeel abundance was moderate, guillemots that were 'off-duty' (not incubating or brooding) spent 478 minutes during daylight resting at the nest site per day. In 1990, when sandeel abundance was extremely low, guillemots that were 'off-duty' spent 28 minutes during daylight resting at the nest site per day. Foraging trips in 1990 lasted more than twice as long as in 1991, although the increased effort was unable to compensate for low food availability and chick mortality at the colony in 1990 (22%) was eleven times higher than in 1991 (Uttley et al. 1994). Smout et al. (2013)

and Kadin et al. (2016) also found that guillemots can increase foraging effort in years of poorer food supply or quality around the colony.

31. For guillemot, Horswill and Robinson (2015) recommend use of baseline age of first breeding at 6 years old, adult (4th year and older) survival 0.939, 2-3 year survival 0.917, 1-2 year survival 0.792 and juvenile (first year) survival 0.56. The much lower survival of juveniles suggests that if there were impacts of displacement from offshore wind farms then those might be most likely to arise among juvenile birds rather than adults. Adult and immature survival rates are influenced by the amount of oil pollution, by weather conditions (survival is lower with warmer sea temperatures and higher winds), and by abundance of prey fish (Harris and Bailey 1992, Sandvik et al. 2005, Votier et al. 2005, 2008).

32. For razorbill, Horswill and Robinson (2015) recommend use of baseline age of first breeding at 5 years old, adult (third year and older) survival 0.895, immature survival (0-2 years) 0.63. The much lower survival of immatures suggests that if there were impacts of displacement from offshore wind farms then those might be most likely to arise among immature (especially juvenile) birds rather than adults. Adult survival rates are influenced by weather conditions (survival is lower with warmer sea temperatures and higher winds), and by abundance of prey fish (Sandvik et al. 2005).

33. The annual mortality of adult guillemots is around 6% per annum and that of adult razorbills is around 10% per annum, and this will include any mortality caused by existing human impacts such as oil pollution, hunting, fishing bycatch and disturbance, as well as 'natural' mortality. Given that all offshore wind farms in UK North Sea waters combined represent an extremely small fraction of potential foraging habitat of guillemots and razorbills within UK North Sea waters, it would seem appropriate to assess the plausible additional mortality caused by offshore wind farm displacement, barrier effects and increased ship traffic as also being extremely small in relation to the total annual mortality, given that this total annual mortality already includes any impact of existing (baseline) human impacts. In that context, to suggest that displacement from an offshore wind farm might increase mortality by 5% or more for all individuals that are displaced seems inconsistent with a total annual mortality of guillemot adults of only 6% or razorbill adults of 10%, when that already includes all impacts from existing human activities throughout the entire year.

What are the likely consequences of displacement for the population?

34. Sutherland (1996) and Newton (1998) pointed out that for migrant birds, such as guillemots and razorbills, population change following habitat loss in their nonbreeding area would depend on the relative strength of density-dependence in

the breeding area and in the nonbreeding area. If the population was regulated by density-dependent competition for breeding resources then habitat loss in the nonbreeding area may be unimportant. Goss-Custard et al. (1997) also pointed out that nonbreeding season habitat loss would only result in a decrease in a waterbird population if the population was subject to density-dependent competition for resources and population size was at carrying capacity of the environment.

35. Evidence strongly indicates that guillemots and razorbills, as other alcids and seabirds in general, are limited by competition for safe breeding sites, either through limitations in food resource surrounding the colony (Crespin et al. 2006, Elliott et al. 2009, Jovani et al. 2016, Sandvik et al. 2016), or through limitation in suitable nest sites (Kokko et al. 2004, Mitchell et al. 2004, Pontier et al. 2008) or a combination of these (Furness and Birkhead 1984, Birkhead and Furness 1985, Wakefield et al. 2017). This would suggest that their population size will be limited by breeding habitat suitability and may not be limited by wintering habitat. Loss of wintering habitat might, therefore, have little or no impact on guillemot or razorbill numbers unless habitat loss was so extensive that nonbreeding season habitat became a limiting factor for the population because their density increased so much that interference competition or prey depletion became a driving factor.

36. Guillemots and razorbills breeding at colonies in the North Sea mostly remain within the North Sea through the nonbreeding period, but these are joined by some birds from Norwegian Sea and Barents Sea colonies (Anker-Nilssen et al. 2000, Wernham et al. 2002, Cherenkov et al. 2016). There are at least 1.5 million guillemots in the North Sea in the nonbreeding period (Blake et al. 1984, Furness 2015). These birds are distributed across the entire North Sea (Camphuysen 2002, MERP), though with slightly higher densities in areas near to colonies and over some shallow sand banks where foraging may be most profitable, and low densities in the small area near Norway where the North Sea is particularly deep (MERP). Guillemots can dive to at least 100 m depth, so can access the sea bed across almost the whole North Sea, but probably prefer to forage in water less than 40 m deep (Daunt et al. 2007). The North Sea is about 750,000 km^2, so the mean density of guillemots is around 2 birds/km^2. This is consistent with the latest mapping of guillemots, which found 2 to 3/km^2 in shallower parts of the North Sea in January, but <1/km^2 in the deepest areas off Norway (MERP). All constructed, consented and proposed offshore wind farms in the North Sea plus 2 km buffers around these sum to approximately 5,000 km^2. If 50% of guillemots were displaced from all offshore wind farms and a buffer area of 2 km around these, and guillemot density was the average of 2 birds per km^2, then 5,000 birds would be displaced. Assuming that all displaced birds remained in the North Sea in places away from offshore wind farms, that would increase the density of guillemots in the rest of the North Sea from 2/km^2 to 2.007/km^2. Even if

offshore wind farms all occupied higher quality habitat so that guillemot density was twice the North Sea average and so twice as many birds were displaced, the density elsewhere would only increase to 2.013/km^2. It is difficult to imagine any biological mechanism that would result in detectable increase in density-dependent competition as a result of such a small increase in mean density of the population. A similar calculation for razorbill would reach this same conclusion. By comparison, the main food fish of guillemots and razorbills fluctuate in abundance much more dramatically. For example, the southern North Sea sandeel has varied between a spawning stock biomass of 76,000 tonnes and 996,000 tonnes (a 13-fold range of densities), while recruitment of young fish in that stock has varied from 25.4 billion to 520.3 billion (a 20-fold range of densities) in different years (ICES 2018). Fluctuations in food abundance seem likely to be much stronger drivers of demography than changes in guillemot and razorbill densities, and that is consistent with the observation that there is a significant relationship between guillemot and razorbill survival rates and food abundance (Sandvik et al. 2005).

37. Probably the most likely consequence is that displacement will have impacts on guillemot and razorbill populations that are too small to detect. Even though there are now many offshore wind farms in the southern North Sea, the total area of these represents a very small fraction of the habitat used by nonbreeding guillemots and razorbills throughout the southern North Sea, so that cumulative habitat loss is very small. The increase in density of guillemots and razorbills caused by displacement away from offshore wind farms will therefore be extremely slight at the regional or biogeographic scale. However, this can be put into some context by looking at the impact of habitat loss for estuarine wader populations that feed on mudflats during the nonbreeding season. Estuarine populations of waders are known to deplete prey in mudflats through the nonbreeding season, and are known to be affected by interference competition. So impacts on estuarine waders are likely to be greater than impacts on populations of birds that are not at the carrying capacity of their nonbreeding habitat.

38. It is known that many shorebirds that feed on mudflats are subject to strong interference competition and prey depletion (Goss-Custard et al. 2006). Estuarine habitat loss caused by barrages at Cardiff Bay and Rhymney resulted in an increase in mortality of 3.17% of displaced redshanks, a species known to be subject to strong density-dependent competition for food in winter due to both prey depletion and interference (Goss-Custard et al. 2006). Oystercatchers are also known to be strongly susceptible to interference competition in winter on tidal mudflats. At Oosterschelde, Netherlands, two-thirds of the tidal mudflat area was destroyed by coastal engineering works (the Delta Works). There was no difference in oystercatcher winter adult survival or in movement rates before and after this

habitat loss, although survival was reduced in severely cold winters compared to mild winters (Duriez et al. 2009). A study of the consequences of saltmarsh habitat loss for individually colour marked dark-bellied Brent geese followed the fate of displaced geese for 13 years after loss of saltmarsh habitat (Ganter et al. 1997). Displaced birds moved more often to less preferred sites that were not filled to capacity than did control birds. However, no significant differences in subsequent survival or fecundity of displaced birds could be found compared to control birds, although there may have been a slight but not statistically significant trend towards displaced birds performing less well than controls (Ganter et al. 1997). The researchers concluded that 'if alternative sites are available there may be no significant fitness consequences to forced dispersal' (i.e. displacement).

39. Based on our understanding of their winter feeding ecology and susceptibility to density-dependent competition, any effect of displacement of guillemots and razorbills would be expected to be less than seen in redshanks, and would be unlikely to be greater than seen in oystercatchers or dark-bellied Brent geese.

40. Despite the uncertainty about impacts on nonbreeding guillemots and razorbills, the available evidence suggests that the most likely result is that there will be little or no impact on adult survival, and that any impact would probably be undetectable at the population level. However, data from further tracking studies, as pioneered by Daunt et al. (2007), on the time-activity budgets of nonbreeding guillemots and razorbills, and on their movements within winters, would allow stronger conclusions to be reached.

41. To conclude this section, we must acknowledge that the impact of displacement of guillemots and razorbills by offshore wind farms is uncertain. However, we do know that natural mortality of adult guillemots and razorbills (including impacts of existing human activities) is very low (6% and 10% per annum respectively), and that displacement of guillemots and razorbills by offshore wind farms is likely to be incomplete, and may reduce with habituation, and that offshore wind farms may in the long term increase food availability to guillemots and razorbills through providing enhanced habitat for fish populations. This suggests that impacts of displacement from offshore wind farms are unlikely to represent levels of mortality anywhere near to the 6% or 10% total annual mortality that occurs due to the combination of many natural factors plus existing human activities. In general, seabirds achieve higher body condition during the non-breeding season than they do while breeding, and the ecology of guillemots and razorbills suggests that density-dependent competition is highly likely during the breeding season, but is less likely to occur in the nonbreeding period. On that basis, it is unlikely that displacement by offshore wind farms would result in an additional mortality exceeding 1% of

displaced birds, and any impact is more likely to be close to zero. Assuming that 1% of displaced birds might die as a consequence of displacement would appear to be highly precautionary. In addition, strong evidence for density-dependent limitation of breeding numbers of guillemots and razorbills suggests that a small increase in winter mortality would have little influence on the size of the guillemot and razorbill populations because they are likely to be at carrying capacity set by breeding habitat suitability.

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Norfolk Vanguard REP7-062: Offshore Ornithology Cumulative and In-combination Collision Risk Assessment Update

Norfolk Vanguard Limited Reference: ExA; AS; 10.D7.21: Cited in this document as Norfolk Vanguard (2019)

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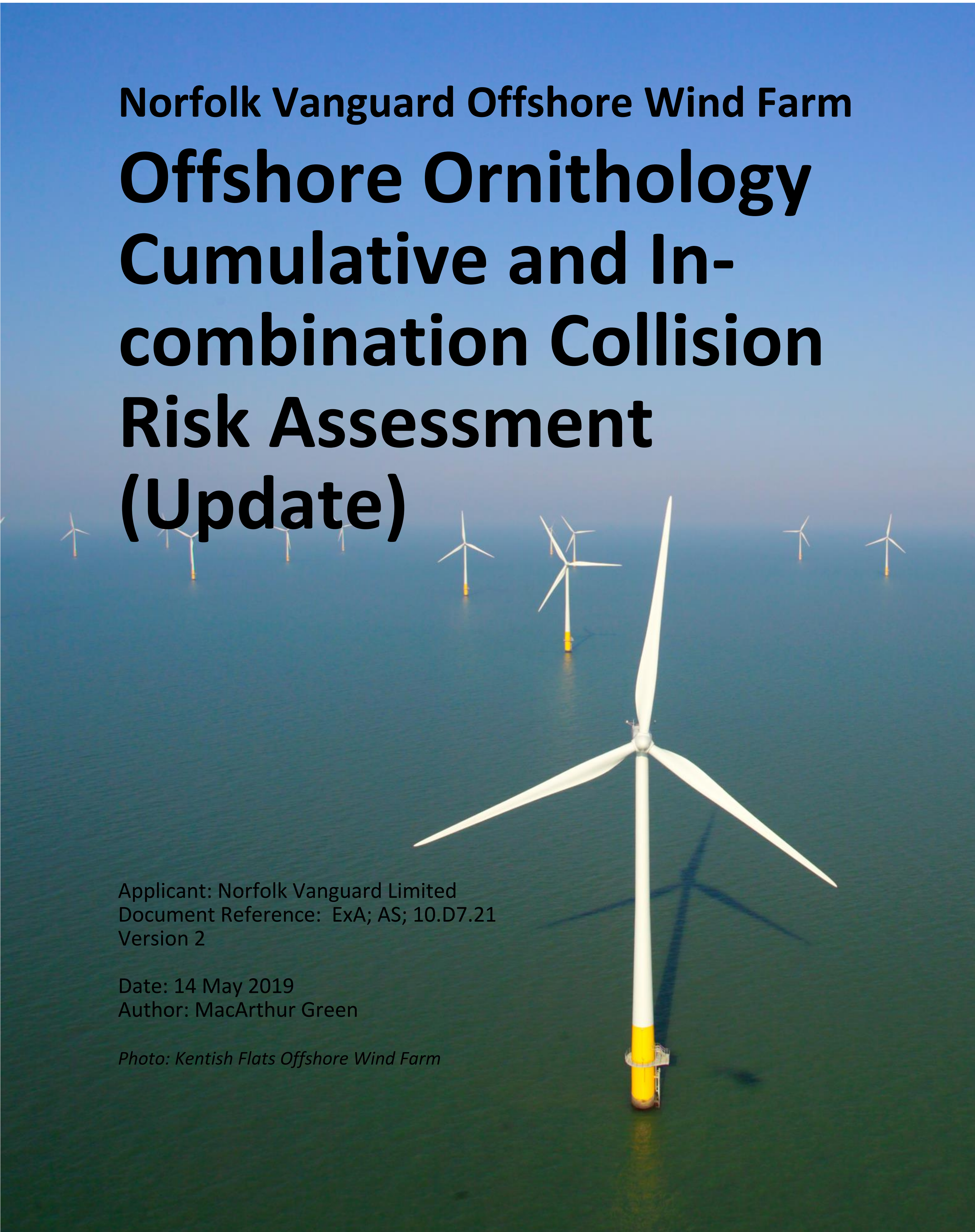

Norfolk Vanguard Offshore Wind Farm

Offshore Ornithology Cumulative and In-combination Collision Risk Assessment (Update)

Applicant: Norfolk Vanguard Limited
Document Reference: ExA; AS; 10.D7.21
Version 2

Date: 14 May 2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
14/05/2019	01D	First for submission	MT	EV	RW

Executive Summary

This note presents an update to the cumulative and in-combination seabird collision risk estimates for the Norfolk Vanguard Offshore Wind Farm (the Project).

Following requests from the Examining Authority (ExA), Natural England and the Royal Society for the Protection of Birds to explore options to mitigate potential seabird impacts from the Project, additional mitigation has been applied through a revision of the wind turbine layout within the offshore sites and an increase in turbine draught height of 5m, from 22m to 27m, to further minimise collision risks.

The revised project design comprises an amendment to the maximum proportion of turbines to be installed across Norfolk Vanguard East and Norfolk Vanguard West. The layout of the wind turbines will be based on the following maxima:

- No more than two-thirds of the turbines will be installed in Norfolk Vanguard West; and
- No more than half of the turbines in Norfolk Vanguard East (with the remainder installed in the other site in each case).

These replace the previous worst case assumption that all of the turbines would be installed in either the Norfolk Vanguard East or Norfolk Vanguard West sites.

The worst case collision prediction for each species for the revised layouts for the Project alone were provided ahead of Issue Specific Hearing 6 in ExA; CRM; 10.D.6.5.1. The average reduction in collision mortality resulting from the revised layouts was 34%, which was in addition to the approximate 10% reduction resulting from the removal of the 9MW turbine from the design envelope. This note provides further updated estimates with additional reduction in collisions obtained through an increase in the turbine draught height (i.e. the gap between the lower rotor tip and the sea surface at Mean High Water Springs) of 5m, from 22m to 27m.

The average collision risk (across species) for the project, accounting for all the design revisions offered as mitigation by the Applicant (i.e. removal of 9MW turbine, revised layout and turbine draught height increase) has been reduced by 65% in comparison to the Environmental Statement as submitted with the Draft Consent Order application in June 2018. The Applicant considers that this represents a significant step forward in considerably reducing the potential collision impacts associated with the Project.

This note provides a summary of the EIA (Environmental Impact Assessment) project alone assessment as provided in ExA; CRM; 10.D.6.5.1 and updated cumulative assessments with the revised Norfolk Vanguard predictions for gannet, kittiwake, lesser black-backed gull, herring gull, great black-backed gull and little gull, and updated HRA (Habitats Regulations

Assessment) in-combination assessments for gannet, kittiwake, lesser black-backed gull and little gull.

The updated assessment concludes that there will be no significant impacts for any species due to cumulative collisions (EIA) and no Adverse Effects on the Integrity of any Special Protection Areas (SPAs) due to collisions for the Project alone or in-combination with other projects (HRA).

Table of Contents

Tables

Glossary

BDMPS	Biologically Defined Minimum Population Scale
CPGR	Counterfactual of Population Growth Rate
CPS	Counterfactual of Population Size
CRM	Collision Risk Model
EIA	Environmental Impact Assessment
ES	Environmental Statement
FFC	Flamborough and Filey Coast
HRA	Habitats Regulations Assessment
JNCC	Joint Nature Conservation Committee
NE	Natural England
MHWS	Mean High Water Springs
NV	Norfolk Vanguard
PCH	Potential Collision Height
PVA	Population Viability Analysis
RSPB	Royal Society for the Protection of Birds
SCM	Seabird Colony Monitoring
SNH	Scottish Natural Heritage
SPA	Special Protection Area

1 INTRODUCTION

1. In the Norfolk Vanguard Deadline 6.5 submission (ExA; CRM; 10.D6.5.1), revised collision estimates were presented for the Project alone to account for the revised project layouts, offered as mitigation for collision risks. This note provides a further update to the collision risk figures following an increase in turbine draught height of 5m (from 22m to 27m). The assessment below includes revised project alone figures and an assessment of the potential collision impacts on seabirds at the proposed Norfolk Vanguard Offshore Wind Farm (the project) alone, cumulatively (Environmental Impact Assessment (EIA)) and in-combination with other projects (Habitat Regulations Assessment (HRA)).

2. Chapter 5 Project Description of the Environmental Statement (ES) provides information on the project design envelope for the wind turbine layout as included in the application. This chapter notes that the detailed design of the layout will be completed during the post-consent phase of the project, however worst case scenarios were assumed for assessments. The worst case scenario in the ES assumed the following maxima:

- 1,800MW in NV East, 0MW in NV West; or
- 0MW in NV East, 1,800MW in NV West.

3. Previous modelling (as presented in ES Chapter 13 Offshore Ornithology and subsequent updates submitted during the examination) presented worst case mortalities estimated in line with these scenarios.

4. During the examination for the project, requests have been made by Natural England and the Royal Society for the Protection of Birds (RSPB) to explore options to mitigate potential seabird impacts from the Project, and these requests have specifically advised that consideration should be given to increasing the turbine draught height.

5. In order to provide additional mitigation with the aim of further minimising collision risk (in accordance with National Policy Statement EN-3 para 2.6.108), the smallest (9MW) turbine was removed from the design envelope, with the 10MW now the smallest under consideration, and subsequently the turbine layout within the site was reviewed. The wind turbine layout is now based between the following maximum proportion of turbines which could be installed in either site with two alternative scenarios, (a) and (b):

   a. The maximum proportion of turbines in NV West would be two-thirds (with one-third in NV East); or

   b. The maximum proportion of turbines in NV East would be half (with the other half in NV West).

6. The above updates were presented in the Deadline 6.5 submission (ExA; CRM; 10.D6.5.1) for the project alone for both scenario (a) and (b) for each species in order to clearly identify the species-specific worst case design, which reflect differences in the densities of a particular species across NV East and NV West. The higher estimate in each case represents the worst case for assessment.

7. In response to Natural England's comments received at Deadline 7, further mitigation to reduce collision risk has been adopted by the Applicant through an increase in the turbine draught height of 5m (from 22m to 27m). Since the density of seabirds in flight decreases with increasing altitude (i.e. most seabirds fly close to the sea), increasing the distance between the rotors and the sea surface reduces collision risk. This reduces the Project collision risk over and above the previous design revisions by 41%.

8. Consequently, in response to requests from Natural England and the RSPB to minimise the project's potential impacts, and in accordance with National Policy Statement EN-3, the Project collision risk has been reduced by 65% since the DCO submission (including the removal of the 9MW turbine, the revised layout and the 5m turbine draught height increase).

9. Natural England provided interim comments on the Applicant's submissions at Deadline 6, some of which are of relevance to this cumulative and in-combination assessment. These are provided in Table 1 and the sections where these have been addressed are identified.

Table 1 Comments provided by Natural England (2019) on the Applicant's Deadline 6 submissions where these are relevant to the current cumulative and in-combination assessment presented in this report.

Comment	Response and section where addressed (if appropriate)
We note that the CRM predictions in the HRA assessments have been adjusted to adult only currency by using the proportion of adults based on the age structure model in BDMPS report (Furness 2015) that was created in order to assess the numbers of immature birds that are associated with breeding populations. We are uncertain as to the appropriateness of assuming that the proportion of adults from this model will be representative of the proportion of adults recorded in the Vanguard areas. As noted in our pre-meeting at the last ISH, we recommend that this would be better undertaken based on the proportion of adults recorded in the baseline survey data for each season from Vanguard, should this be available.	The survey derived age ratios have been reviewed and for all relevant species (gannet, kittiwake and lesser black-backed gull) these are in excess of 93%. The Applicant does not consider these to provide a reliable guide for use in the assessment. Further consideration of the evidence for at sea age ratios will be provided in a subsequent submission.
Baseline mortality rates for HRA assessments have been based on using an all ages colony count and all ages survival/mortality rate to calculate baseline mortality. We note that in our Relevant	Additional assessment is provided in this note which considers the proportion of total collisions

Comment	Response and section where addressed (if appropriate)
Representations, which is actually repeated by the Applicant here in Table 1 of this document that: 'Given that the outputs of the existing PVAs tend to be on an adult currency, we also advise that calculations of baseline mortality used in the HRA are undertaken on an adult currency, therefore using the adult colony figure and the adult mortality rate rather than on all ages.' Therefore, we advise again that assessments should be done using baseline mortality calculations using the adult colony figures and adult mortality rates.	assigned to adults (using the Furness 2015 age ratios) assessed against the SPA populations using the adult mortality rates. The Applicant would note that the assessments against background mortality are also supplemented with assessment using PVA (Population Viability Analysis), and consideration of these results is the primary basis for conclusions reached.
We welcome that in-combination assessments have been undertaken including Hornsea Three and excluding Hornsea Three. As previously noted, the latest figures available from the Thanet Extension and Hornsea Project Three examinations should be presented in the in-combination assessment, and the significant lack of confidence regarding the Hornsea Project Three figures should be discussed.	The cumulative and in-combination assessments in this note use figures for Hornsea Project Three (from the Environmental Statement, ES) and Thanet Extension (from that Project's Deadline 3 submission, Appendix 39), based on the advice received from Natural England. This was on the basis that Natural England could not advise on which alternative assessment values submitted during each project's examination were appropriate. The current assessment therefore also presents the ES values for these projects.
We welcome that the in-combination assessments for gannet and kittiwake at the FFC SPA now include figures for the Hywind, Kincardine and Moray West offshore wind farms (OWFs). We also note that the CRM figures included in the in-combination assessments for East Anglia One are the figures for the 150 turbine option (which is the legally secured design). However, we note that the in-combination assessment for LBBG at the Alde-Ore Estuary SPA still does not include figures for the Hywind and Kincardine OWFs. We will review the in-combination assessments for the other species regarding these other OWFs for Deadline 7.	The Applicant acknowledges that collision estimates for lesser black-backed gull at Hywind and Kinkardine wind farms were omitted from the assessment. These have now been included within this note, although it should be noted that no collisions were predicted for this species at either wind farm.
We note that there are no updated assessments of EIA cumulative gannet, kittiwake and LBBG CRM presented in the relevant assessment sections for these species. Whilst the updated cumulative CRM figures are presented in Table 4 for gannet and Table 13 for kittiwake, the cumulative totals and hence the significance of cumulative CRM impacts have not yet been agreed and therefore the Applicant should also present the updated assessment of what these cumulative figures equate to of baseline mortality of the largest BDMPS and biogeographic populations. Updated cumulative CRM assessments should be presented for all of the five key species (gannet, kittiwake, LBBG, herring gull and great black-backed gull).	These have been provided in this note: gannet cumulative (section 3.1.1.3), kittiwake cumulative (section 3.2.1.3), lesser black-backed gull cumulative (section 3.4.1.4), herring gull cumulative (section 3.3.1.2) and great black-backed gull cumulative (section 3.5.1.2).
We note that no updated assessment is provided for great black-backed gull (GBBG) – we advise that as no agreements have yet been made regarding GBBG cumulative CRM, that an updated	This is provided in this note (section 3.5.1.2).

Comment	Response and section where addressed (if appropriate)
assessment should also be provided that takes account of the updated figures for Vanguard, Thanet Extension and Hornsea Three, and also includes figures for Hywind, Kincardine and Moray West OWFs.	

10. Natural England's responses submitted at Deadline 7 included a request to investigate the potential for further reducing collision mortality through raising rotor blade heights. This note, together with ExA; As; D7.5.2, provides updated collision risk modelling results and assessment of impacts which directly address this request.

2 OVER PRECAUTION

11. In the following assessment, it is considered that many sources of precaution have been applied. These are highlighted by the Applicant within the relevant species specific sections. It should also be noted that as the number of wind farms included in cumulative and in-combination assessments increases, the total predicted impacts also increase. The various sources of precaution which have become accepted components of ornithological assessments are therefore highlighted. Under the current methodologies, the Applicant considers that there is a very real risk that future wind farms may face a conclusion of adverse effect on integrity being reached, not because they represent unacceptable ecological impacts at a project alone or in-combination level, but because the assessment process does not account for the need to present and manage uncertainty in a proportionate manner.

12. The sources of precaution in ornithological impact assessments include reduced collision risks for built wind farms compared with their consented designs which are considered in the assessments, the use of upper confidence limits and overly precautionary parameter estimates in collision risk modelling and displacement assessments and a preference for density independent population models, despite the fact that these are self-evidently flawed for predictive purposes. The combined result of these and other precautionary assumptions means that cumulative impact estimates are almost certainly overinflated, possibly by several orders of magnitude.

3 UPDATED ASSESSMENT

3.1 Gannet

3.1.1 Collision risk

3.1.1.1 EIA Project alone

13. The revised collision risk estimates for gannet for the 10MW turbine, along with the revised Project layout (ExA; CRM 10.D6.5.1) and 5m turbine draught height increase (ExA;AS;10.D7.5.2), calculated using the Band (2012) deterministic model and Natural England's preferred parameter values, are provided in Table 2.

Table 2. Gannet seasonal and annual collision risk using the migration free (April to August) and full (March to September) breeding seasons.

Breeding season	Spring	Migration free breeding	Autumn	Annual
Migration-free	11.88 (6.64-19.04)	11.98 (2.3-25.21)	42.45 (28.27-59.2)	66.31 (37.21-103.44)
Full	10.89 (6.64-16.51)	16.98 (3.8-35.15)	38.43 (26.77-51.77)	

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015).

14. In the submission at Deadline 6.5 (ExA; CRM 10.D6.5.1, Table 2) it was concluded that a higher annual mortality of 112 (prior to the turbine draught height increase) would not increase the background rate by more than 1% and therefore it was concluded that Norfolk Vanguard Offshore Wind Farm alone would have no significant impact at the EIA scale. This conclusion is further supported by the lower revised annual estimate of 66 (a reduction of 41% for the turbine draught height increase alone), and therefore the conclusion of no significant impact at the EIA scale remains valid.

3.1.1.2 HRA Project alone

15. The proportion of the total collisions assigned to the Flamborough and Filey Coast (FFC) SPA in each season by the Applicant in the Information to Support HRA submitted with the Application (document reference 5.03) were 100% (breeding season), 4.2% (autumn) and 5.6% (spring). These rates were derived using the population estimates in Furness (2015) and evidence derived from tracking studies on the migration routes taken by birds from UK colonies (see Norfolk Vanguard 2019a).

16. Natural England (2018) advised the Applicant that the nonbreeding season rates should only account for the relative population sizes, with recommended rates calculated by Natural England of 4.8% in autumn and 6.2% in spring (Schedule of

Natural England's responses to Examining Authority's second round of written questions, 13 March 2019). Both sets of rates (the Applicant's original set and Natural England's preferred set) have been used to estimate the number of predicted collisions at Norfolk Vanguard which would be attributed to the Flamborough and Filey Coast SPA population (Table 3), using the revised worst case Norfolk Vanguard estimates (the worst case for gannet assumes 50% of the turbines are located in Norfolk Vanguard East and 50% are located in Norfolk Vanguard West) and the reduction due to the 5m turbine draught height increase.

Table 3. Gannet seasonal and annual collision risk apportioned to the Flamborough and Filey Coast SPA using the migration free (April to August) and full (March to September) breeding seasons.

Breeding season	Apportioning rates	Spring	Breeding	Autumn	Annual
Migration-free	Applicant	0.5	12.0	2.4	14.9
	Natural England	0.6	12.0	2.6	15.2
Full	Applicant	0.5	17.0	2.2	19.6
	Natural England	0.5	17.0	2.4	19.9

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015). Only the worst case estimates for Norfolk Vanguard East are shown.

17. The maximum predicted mortality of Flamborough and Filey coast SPA gannets at Norfolk Vanguard, using the full breeding season and Natural England's preferred apportioning rates is 19.9 adults (95% confidence intervals 5.8-39.2).

18. The SPA population at designation was 11,061 pairs (22,122 individuals, although this had increased to 13,391 pairs by 2017). At an average natural adult mortality rate of 0.081, the natural annual mortality of the population is 1,792 (designated) to 2,169 (2017 count). The addition of 19.9 individuals would therefore increase the mortality rate by 1.1% (designated) and 0.9% (2017 count). If the upper 95% confidence estimate (39.2) is used, these increases would be between 2.2% and 1.8%, respectively. While if the lower 95% confidence estimates are used (5.8) these rates are 0.3% and 0.3%.

19. While the mean prediction using the designated population is slightly above the 1% threshold for detection, with the consequent need to undertake additional assessment, it is important to note that this collision prediction combines several sources of precaution:

- Use of a nocturnal activity rate of 25% (Furness et al. 2018 recommended this should be 8% in the breeding season and 4% in the nonbreeding season);
- Assignment of all collisions between March and September (the full breeding season) to the SPA makes no allowance for the presence of immature birds from

a wide range of other colonies which are likely to be present at this time, or for the presence of late and early migrants, and;

- Bowgen and Cook (2018) recently estimated a gannet collision avoidance rate from an empirical study of 99.5%, which would more than halve the estimates above calculated using 98.9%.

20. Outputs from a PVA model for this population were presented for Hornsea Project Three (MacArthur Green 2018). This model was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. Outputs from this model were presented as additional adult mortality at increments of 25, thus the results for additional adult mortality of 25 and 50, the closest values to the current predictions of 5.8, 19.9 and 39.2 are provided in Table 4.

Table 4. Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
		Growth rate	Population size	
Rate set 1, density independent	25	0.999	0.968	Table A2 1.1 & 1.3
	50	0.998	0.937	
Rate set 1, density dependent	25	0.999	0.978	Table A2 2.1 & 2.3
	50	0.999	0.957	
Rate set 2, density independent	25	0.999	0.968	Table A2 3.1 & 3.3
	50	0.998	0.936	
Rate set 2, density dependent	25	0.999	0.978	Table A2 4.1 & 4.3
	50	0.999	0.957	

21. The maximum reduction in the population growth rate, at an adult mortality of 50, using the most precautionary combination of assumptions (95% confidence estimate, all mortality assigned to adults, assessed using the density independent model) was 0.2% (0.998). Using the more realistic density dependent model the maximum reduction in growth rate was 0.1% (0.999).

22. These compare to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. A reduction of no more than 0.2% (and that for a considerably higher mortality than even the most precautionary assumption using the upper 95% confidence estimate) in this growth rate represents a negligible risk for the population.

23. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

24. On the basis of the population model predictions the number of predicted project alone gannet collisions attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a slight reduction in the growth rate currently seen at this colony, and so would not have an adverse effect on integrity of the SPA.

25. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from collision impacts on gannet due to the proposed Norfolk Vanguard project alone.

3.1.1.3 EIA cumulative and HRA In-combination

26. Natural England advised that the cumulative and in-combination collision assessment should include estimates for three additional Scottish wind farms (Hywind, Kincardine and Moray West) and that there is uncertainty regarding the appropriate values to use for the Hornsea Project Three and Thanet Extension wind farms as these are also currently in examination and therefore there is potential for variation. Following the Applicant’s understanding from discussions with Natural England, values for Thanet Extension were obtained from the Thanet Extension submission at Deadline 3 (Appendix 39) and estimates for Hornsea Project Three have been taken from the project’s ES. As set out above, in accordance with Natural England's advice, cumulative totals without Hornsea Project THREE are also provided. Table 5 presents the full updated cumulative and in-combination predictions.

Table 5. Gannet collision mortality for all wind farms, and with collisions apportioned to the Flamborough and Filey Coast SPA

Wind farm	Spring		Breeding		Autumn		Annual	
	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
Beatrice Demonstrator	0.7	0.05	0.6	0.0	0.9	0.04	2.2	0.1
Greater Gabbard	4.8	0.30	14.0	0.0	8.8	0.42	27.5	0.7
Gunfleet Sands	0.0	0.00	0.0	0.0	0.0	0.00	0.0	0.0
Kentish Flats	1.1	0.07	1.4	0.0	0.8	0.04	3.3	0.1
Lincs	1.7	0.10	2.1	2.1	1.3	0.06	5.0	2.3
London Array	1.8	0.11	2.3	0.0	1.4	0.07	5.5	0.2
Lynn and Inner Dowsing	0.2	0.01	0.2	0.2	0.1	0.01	0.5	0.2
Scroby Sands	0.0	0.00	0.0	0.0	0.0	0.00	0.0	0.0
Sheringham Shoal	0.0	0.00	14.1	14.1	3.5	0.17	17.6	14.3

Wind farm	Spring		Breeding		Autumn		Annual	
	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
Teesside	0.0	0.00	4.9	2.4	1.7	0.08	6.7	2.5
Thanet	0.0	0.00	1.1	0.0	0.0	0.00	1.1	0.0
Humber Gateway	1.5	0.09	1.9	1.9	1.1	0.05	4.5	2.0
Westermost Rough	0.2	0.01	0.2	0.2	0.1	0.01	0.5	0.2
Hywind	0.8	0.05	5.6	0.0	0.8	0.04	7.2	0.1
Kincardine	0.0	0.00	3.0	0.0	0.0	0.00	3.0	0.0
Beatrice	9.5	0.59	37.4	0.0	48.8	2.34	95.7	2.9
Dudgeon	19.1	1.18	22.3	22.3	38.9	1.87	80.3	25.3
Galloper	12.6	0.78	18.1	0.0	30.9	1.48	61.6	2.3
Race Bank	4.1	0.25	33.7	33.7	11.7	0.56	49.5	34.5
Rampion	2.1	0.13	36.2	0.0	63.5	3.05	101.8	3.2
Hornsea Project One	22.5	1.40	11.5	11.5	32.0	1.54	66.0	14.4
Blyth Demonstration Project	2.8	0.17	3.5	0.0	2.1	0.10	8.4	0.3
Dogger Bank Creyke Beck Projects A and B	4.3	0.27	5.6	2.8	6.6	0.32	16.5	3.4
East Anglia ONE	6.3	0.39	3.4	3.4	131.0	6.29	140.7	10.1
European Offshore Wind Deployment Centre	0.1	0.00	4.2	0.0	5.1	0.25	9.3	0.3
Firth of Forth Alpha and Bravo	65.8	4.08	800.8	0.0	49.3	2.37	915.9	6.4
Inch Cape	5.2	0.32	336.9	0.0	29.2	1.40	371.3	1.7
Moray Firth (EDA)	8.9	0.55	80.6	0.0	35.4	1.70	124.9	2.3
Neart na Gaoithe	23.0	1.43	143.0	0.0	47.0	2.26	213.0	3.7
Dogger Bank Teesside Projects A and B	10.8	0.67	14.8	7.4	10.1	0.49	35.7	8.5
Triton Knoll	30.1	1.87	26.8	26.8	64.1	3.08	121.0	31.7
Hornsea Project Two	6.0	0.37	7.0	7.0	14.0	0.67	27.0	8.0
East Anglia THREE	9.6	0.60	6.1	6.1	33.3	1.60	49.0	8.3
Hornsea Project Three	8.0	0.45	18.0	18.0	12.0	0.5	38.0	19.0
Thanet Extension	22.9	1.42	0.0	0.0	11.1	0.53	34.0	2.0
Moray West	1.0	0.06	10.0	0.0	2.0	0.10	13.0	0.2
Norfolk Vanguard	10.9	0.5	16.9	16.9	38.4	2.4	66.3	19.9
Total (inc. Hornsea Project Three)	**298.4**	**18.27**	**1688.2**	**176.8**	**737**	**35.89**	**2723.5**	**231.1**
Total (exc. Hornea Project Three)	**290.4**	**17.82**	**1670.2**	**158.8**	**725**	**35.39**	**2685.5**	**212.1**

3.1.1.3.1 Cumulative assessment

27. The cumulative total, all age class annual gannet collision mortality is estimated to be 2,723.5 with the inclusion of Hornsea Project Three and 2,685.5 without this project. Note, however that many of the collision estimates for other wind farms were calculated on the basis of consented designs; that is with higher total rotor

swept areas than have been installed (or are planned), which is a key factor in collision risk. For example, the Beatrice wind farm, which is currently under construction, was consented on the basis of up to 125 x 7MW turbines but only 84 x 7MW turbines have been installed, leading to a reduction in mortality risk of 33%. A method for updating collision estimates for changes in wind farm design such as this was presented in Trinder (2017). Updating the collision estimates for the Beatrice wind farm using this approach reduces the predicted annual mortality from 96 to 64. Applying the same method to the other relevant wind farms achieves a reduction in the cumulative annual mortality of around 400. Therefore, the values presented in Table 5, as well as being based on precautionary calculation methods, can be seen to overestimate the total collision risk by around 15% due to the reduced collision risks for projects which undergo design revisions post-consent.

28. Previous gannet collision assessments for the wind farms listed in Table 5 have been made on the basis of Band model Option 1 and a range of avoidance rates between 95% and 99%. The current rate of 98.9% dates from November 2014 (JNCC *et al.,* 2014) and followed the review conducted by Cook *et al.* (2014). Therefore, the decisions for some of the projects consented prior to this date were on the basis of estimated cumulative collision mortality numbers which were higher than the values presented in Table 5. However, given the variation in rates presented in different assessments and the rates used in reaching consent decisions, it is difficult to confidently determine the avoidance rate used for each wind farm consent decision. Nonetheless, it can be stated with a good degree of certainty that none of the previous wind farms have been consented on the basis of an avoidance rate higher than 99%, and many will have been based on assessment at 98%. A reduction in the avoidance rate from 99% to 98% leads to a doubling of the predicted collisions, therefore even though cumulative totals for older wind farms included fewer wind farms this will have been more than offset by the lower avoidance rate used.

29. Therefore, since avoidance rates have such a large effect on predicted mortality levels it therefore follows that the current cumulative total of 2,723.5 is almost certainly lower than those calculated for previous wind farm cumulative assessments (for which consent decisions have been granted).

30. Work conducted at the Greater Gabbard wind farm (APEM, 2014) has also found that gannet avoidance of wind farms during the autumn migration period may be even higher than the current estimate of 98.9%. Of 336 gannets observed during this study, only 8 were recorded within the wind farm, indicating a high degree of wind farm (macro) avoidance. Analysis of their data indicated a macro-avoidance rate in excess of 95% compared with the current guidance value of 64%. When combined with meso- and micro-avoidance this would result in higher overall

avoidance than the current 98.9% and would further reduce the total collision mortality prediction.

31. Furthermore, the collision estimates for most wind farms have used a nocturnal activity rate for this species of 25% in all months, which is much higher than those identified from analysis of tag data for the breeding and nonbreeding seasons (8% and 4%; Furness et al. 2018). It is straightforward to adjust existing mortality estimates to account for this reduction (i.e. it is not necessary to rerun the collision model for this update). However, it is necessary to calculate a mortality adjustment rate for each month at each wind farm because the duration of night varies with month and latitude (both of which are inputs to the collision model). This has not been undertaken for the current assessment but would be expected to reduce the cumulative total by at least 10%. This further emphasises the precautionary nature of the current assessment.

32. The background mortality for the BDMPS population (456,298), using an all age mortality rate, is 87,153, and for the biogeographic population (1,180,000) is 225,380. An addition of 2,723.5 to this increases the mortality rate by 3.1% (BDMPS) and 1.2% (biogeographic). As these are above the 1% level considered to be the threshold for detectability, further consideration of this is provided below.

33. Demographic data were collated for the British gannet population to produce a population model which was used to consider the potential impact of additional mortality (WWT, 2012). Two versions of the model were developed, with and without density dependence. Of these two models, the density independent one was considered to provide more reliable predictions since it predicted baseline growth at a rate close to that recently observed (1.28% per year compared with an observed rate of 1.33%) while the density dependent model predicted baseline growth of 0.9%.

34. The study concluded that, using the density independent model, on average population growth would remain positive until additional mortality exceeded 10,000 individuals per year, while the lower 95% confidence interval on population growth remained positive until additional mortality exceeded 3,500 individuals, which is greater than the cumulative total in Table 5. Consideration was also given to the risk of population decline. The risk of a 5% population decline was less than 5% for additional annual mortalities below 5,000 (using either the density dependent or density independent model; WWT, 2012).

35. It is important to note that the gannet model presented in WWT (2012) was based on the whole British population, so collisions at wind farms on the west coast (e.g. Irish Sea) also need to be added for consistency. However, a review of applications

in the Irish Sea and Solway Firth (Barrow, Burbo Bank, Burbo Bank Extension, Gwynt Y Mor, North Hoyle, Ormonde, Rhyl Flats, Robin Rigg, Walney 1 and 2, Walney Extension and West of Duddon Sands) gave a gannet annual collision cumulative total of 32.4 at an avoidance rate of 98.9%. Therefore, inclusion of these wind farms in the assessment does not alter the conclusion that cumulative collisions are below a level at which a significant impact on the British gannet population would result.

36. Furthermore, the WWT (2012) analysis was conducted using the estimated gannet population in 2004 (the most recent census available at that time), when the British population was estimated to be 261,000 breeding pairs. The most recent census indicates the equivalent number of breeding pairs is now a third higher at 349,498 (Murray *et al.,* 2015). This increase in size will raise the thresholds at which impacts would be predicted and therefore further reduces the risk of significant impacts.

37. In conclusion, the cumulative impact on the gannet population due to collisions is considered to be of low magnitude, and the relative contribution of the proposed Norfolk Vanguard project to this cumulative total is very small. Gannet is considered to be of low to medium sensitivity to collision mortality and the impact significance is therefore **minor adverse**.

3.1.1.3.2 In-combination assessment

38. The in-combination total, adult annual gannet collision estimate for the FFC SPA is 231, of which Norfolk Vanguard contributes 19.9 (although it should be noted that this is considered to be an over-estimate due to the precautionary assumptions noted above). The in-combination total annual gannet collision estimate, without Hornsea Project Three, is 212.

39. The increase in the background mortality due to this in-combination collision risk (including Hornsea Project Three) is between 12.9% (designated population) and 10.6% (2017 count). Without Hornsea Project Three these increases are 11.8% and 9.8%, respectively.

40. Outputs from a PVA model for this population were presented for Hornsea Project Three (MacArthur Green 2018). This model was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (up to 35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for adult mortality levels of 225 and 250 (the values which most closely correspond to the above estimates) are provided in Table 6.

Table 6. Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
		Growth rate	Population size	
Rate set 1, density independent	225	0.990	0.743	Table A2 1.1 & 1.3
	250	0.989	0.719	
Rate set 1, density dependent	225	0.994	0.814	Table A2 2.1 & 2.3
	250	0.993	0.796	
Rate set 2, density independent	225	0.990	0.743	Table A2 3.1 & 3.3
	250	0.989	0.719	
Rate set 2, density dependent	225	0.994	0.814	Table A2 4.1 & 4.3
	250	0.993	0.795	

41. The maximum reduction in the population growth rate, at an adult mortality of 250, using the more precautionary density independent model was 1.1% (0.989). Using the more realistic density dependent model the maximum reduction in growth rate was 0.7% (0.993).

42. These compare to the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year. A reduction of just over 1% in this case represents a negligible risk for the population.

43. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

44. On the basis of the population model predictions the number of predicted in-combination gannet collisions attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a slight reduction in the growth rate currently seen at this colony, and so would not have an adverse effect on the integrity of the SPA.

45. These totals also include several sources of precaution, including (among other sources of precaution) over-estimated nocturnal activity for existing projects and the use of consented collision estimates for projects which have since been constructed to designs with much lower collision risks.

46. Therefore, the conclusions presented in the Norfolk Vanguard ES and HRA and subsequent submissions (ExA; AS; 10.D6.17) remain the same; it can be concluded that there will be no adverse effect on the integrity of the Flamborough & Filey Coast

SPA from collision impacts on gannet due to the proposed Norfolk Vanguard project in-combination with other plans and projects.

47. Furthermore, the contribution from Norfolk Vanguard to this total has been substantially reduced following design revisions to mitigate collision risks, with the annual collision mortality estimate for gannet reduced by 66% when the removal of the 9MW turbine, revised layout and turbine draught height increase are considered together.).

3.1.2 Combined displacement and collision risk

3.1.2.1 HRA In-combination

48. Adding the in-combination annual gannet collision estimate of 231 (estimated using Natural England's preferred methods and including Hornsea Project Three) to the in-combination annual displacement prediction of 49 to 65 (see section 2.1.2 of ExA; AS; 10.D6.17), gives a combined SPA mortality estimate of 280 to 296. It is important to note that, on top of the precaution in the individual collision and displacement assessments, summing these two impacts adds another layer of precaution, since it implies that individuals can both be displaced (and suffer increased mortality as a consequence) and also be at risk of collision mortality.

49. However, the above over-precaution notwithstanding, the increase in the background mortality of the SPA population due to this combined in-combination collision and displacement risk was between 15.6% and 16.5% (designated population) and 12.9% and 13.6% (2017 count).

50. Outputs from the gannet PVA model for this population (MacArthur Green 2018) for adult mortality levels of 275 and 300 (the nearest values to this impact prediction) are provided in Table 7.

Table 7. Gannet FFC SPA population modelling results from MacArthur Green (2018).

Model	Adult mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
		Growth rate	Population size	
Rate set 1, density independent	275	0.988	0.699	Table A2 1.1 & 1.3
	300	0.986	0.673	
Rate set 1, density dependent	275	0.992	0.776	Table A2 2.1 & 2.3
	300	0.991	0.757	
Rate set 2, density independent	275	0.988	0.696	Table A2 3.1 & 3.3
	300	0.986	0.673	
Rate set 2, density dependent	275	0.992	0.776	Table A2 4.1 & 4.3
	300	0.991	0.757	

51. The maximum reduction in the population growth rate, at a mortality of 300, using the more precautionary density independent model was 1.4% (0.986). Using the more realistic density dependent model the maximum reduction in growth rate was 0.9% (0.991).

52. On the basis of the observed rate at which this population has grown over the last 25 years, which has been at least 10% per year, a maximum reduction of 1.4% to this rate represents a negligible risk for the population.

53. The gannet breeding numbers at the Flamborough and Filey Coast SPA have continued to increase in all counts conducted to date (most recent 2017) and the gannet population is therefore clearly in favourable conservation status. The relevant conservation objective is to maintain favourable conservation status of the gannet population, subject to natural change.

54. On the basis of the population model predictions the number of predicted in-combination gannet collisions and mortality due to displacement attributed to the Flamborough & Filey Coast SPA is not at a level which would trigger a risk of population decline, but would only result in a slight reduction in the growth rate currently seen at this colony, and so would not have an adverse effect on integrity of the SPA.

55. These totals also include several sources of precaution, including over-estimated nocturnal activity for existing projects and the use of consented collision estimates for projects which have since been constructed to designs with much lower collision risks.

56. Therefore, it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from impacts on gannet due to the proposed Norfolk Vanguard project in-combination with other plans and projects.

3.2 Kittiwake

3.2.1 Collision risk

3.2.1.1 EIA Project alone

57. The revised collision risk estimates for kittiwake for the 10MW turbine and the revised project layout (ExA; CRM 10.D6.5.1) and 5m turbine draught height increase (ExA;AS;10.D.7.5.2), calculated using the Band (2012) deterministic model and Natural England’s preferred parameter values, are provided in Table 8.

Table 8. Kittiwake seasonal and annual collision risk using the migration free (April to August) and full (March to September) breeding seasons.

Breeding season	Spring	Migration free breeding	Autumn	Annual
Migration-free	55.9 (38.49-76.41)	26.58 (9.89-48.14)	32.92 (18.53-50.2)	115.4 (66.9-174.75)
Full	38.66 (27.17-52.26)	45.26 (21.41-75.45)	31.48 (18.32-47.03)	

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015).

58. In the submission at Deadline 6.5 (ExA; CRM 10.D6.5.1, Table 2) it was concluded that a higher annual mortality of 186 (prior to the turbine draught height increase) would not increase the background rate by more than 1% and therefore the Norfolk Vanguard Offshore Wind Farm alone would have no significant impact at the EIA scale. This conclusion is further supported by the lower revised annual estimate of 115 (a reduction of 38% for the turbine draught height increase alone), and therefore the conclusion of no significant impact at the EIA scale remains valid.

3.2.1.2 HRA Project alone

59. The revised total collision risks for kittiwake, calculated using the Band (2012) deterministic model and Natural England's preferred parameter values are provided in Table 8. The proportion of the total collisions assigned to the Flamborough and Filey Coast SPA in each season by the Applicant in the original HRA (Vattenfall 2018) were 16.8% (breeding season), 5.4% (autumn) and 7.2% (spring). These rates were derived using the population estimates in Furness (2015; see Norfolk Vanguard, 2019b).

60. Natural England advised the Applicant (Natural England 2018) that the breeding season rate should take account of more recent tracing studies (Wischnewski 2018) which had found evidence to indicate that the previously accepted foraging range for this species may have been an underestimate.

61. The study's authors (the Royal Society for the Protection of Birds, RSPB) provided the tracking data on request in order to enable analysis to estimate an alternative breeding season apportioning rate.

62. In the 2017 breeding season this research project successfully tracked 18 kittiwakes for periods of up to 29 days in June and July. A summary of the relevant foraging distances recorded by this study is provided below. Of relevance to this analysis are the distances from the SPA to the Norfolk Vanguard sites (205 km to Norfolk Vanguard West and 233 km to Norfolk Vanguard East):

- In June, 12 of 17 birds tracked (in this month) had maximum foraging ranges less than 205 km and 16 had ranges less than 233 km.

- In July, 5 of 11 birds tracked (in this month) had maximum foraging ranges less than 205 km and 7 had ranges less than 233 km.

63. These data indicate that earlier in the season (June) very few birds travelled as far as Norfolk Vanguard and that, even later in the season, foraging trips extending as far as Norfolk Vanguard were only undertaken by around half the tagged birds.

64. It is important that these results are not over-interpreted, since they represent a single season and only a small number of individuals. Nevertheless, they suggest that there is likely to be connectivity between the Flamborough and Filey Coast SPA and Norfolk Vanguard in the breeding season, albeit this connectivity is probably quite low.

65. While some birds recorded on Norfolk Vanguard in the breeding season are therefore likely to have come from Flamborough and Filey Coast SPA, there remains the question of the likely origin of other birds on the site. Immature kittiwakes tend to remain in overwintering areas longer into the breeding season and to move more slowly back towards their natal colonies, both within years and also as they approach maturity (Coulson 2011). Thus, one approach to estimating the kittiwake population size in the North Sea in the breeding season is to consider the spring season immature population in this region, on the basis that these birds are more likely to remain in this area.

66. The UK North Sea spring migration BDMPS immature population is 252,001 (Furness 2015). If this is assumed to represent the UK North Sea population of nonbreeding birds during the breeding season, then this suggests that the Flamborough and Filey Coast SPA adult population (89,040) would make up 26.1% of the birds that could be recorded on Norfolk Vanguard (89040/(252,001+89040)). While it is likely that not all of these immatures would be present in the southern North Sea throughout the breeding season, this figure (252,001) does not include any immature birds from the very large Russian and Norwegian populations. If these birds (1,830,400 immatures) are added to the potential North Sea population the percentage attributed to the SPA is reduced to 4.1% (89040/(89040+252001+1,830,400)). This figure of 4.1% provides a lower value to balance against what is likely to be an upper estimate of 26.1% calculated without these birds. It is acknowledged that in calculating the lower estimate the number of Russian and Norwegian immatures present in the North Sea is almost certainly over-estimated, but it does indicate that the real value is likely to be between 4.1% and 26.1%, and a very substantial number of Russian and Norwegian immature birds are very likely to be present in the southern North Sea.

67. Furthermore, immature birds tend to be less competitive than breeding adults, therefore as distance from colonies increases, the likelihood that birds encountered

are sub-dominant immature individuals increases. Hence the range 4.1% to 26.1% is considered to provide a realistic range of the apportioning rates for FFC SPA birds on Norfolk Vanguard, covering the uncertainty in this calculation. Taking a precautionary approach, it has been assumed that the upper value (26.1%) is applicable to Norfolk Vanguard.

68. This estimated rate was presented to Natural England and the RSPB and discussed during a call on the 2nd April 2019. In their Deadline 7 submission, Natural England advised the Applicant that they should give consideration to a wider range of possible breeding season connectivity percentages, including up to 100% (i.e. all birds on Norfolk Vanguard during the breeding season should be treated as breeding adults from the SPA, although Natural England acknowledged this figure was highly precautionary and unrealistic). The Applicant considers such an approach is extremely precautionary and gives undue weight to the single tagging study conducted in 2017. Further consideration of the kittiwake data has been undertaken and is presented in the following paragraphs. In addition, a review of evidence on kittiwake movements throughout the year in relation to age classes and colonies is being prepared and will be submitted by the Applicant at Deadline 8.

69. Table 9 provides monthly and seasonal kittiwake collision estimates on Norfolk Vanguard for the two alternative development scenarios (a and b, equating to splits across the West and East sites of (a) two-thirds and one-third and (b) half in each), for both the migration free and full breeding seasons (and with the reduction due to increased turbine draught height).

Table 9. Kittiwake monthly collision risks on Norfolk Vanguard with migration free (May to July) and full (March to August) breeding seasons indicated. Scenario (a) corresponds to two-thirds of the turbines in Norfolk Vanguard West and one-third in Norfolk Vanguard East and scenario (b) corresponds to half in each site.

<table>
<tr><th rowspan="3">Month</th><th colspan="2">Monthly</th><th colspan="4">Seasonal total</th></tr>
<tr><th colspan="2"></th><th colspan="2">Migration free</th><th colspan="2">Full breeding</th></tr>
<tr><th>Scenario a</th><th>Scenario b</th><th>Scenario a</th><th>Scenario b</th><th>Scenario a</th><th>Scenario b</th></tr>
<tr><td>Jan</td><td>18.8</td><td>26.9</td><td rowspan="4">46.7</td><td rowspan="4">64.1</td><td rowspan="2">27.2</td><td rowspan="2">38.7</td></tr>
<tr><td>Feb</td><td>8.5</td><td>11.8</td></tr>
<tr><td>Mar</td><td>13.3</td><td>17.2</td><td rowspan="6">39.8</td><td rowspan="6">45.3</td></tr>
<tr><td>Apr</td><td>6.1</td><td>8.2</td></tr>
<tr><td>May</td><td>6.0</td><td>7.8</td><td rowspan="3">16.2</td><td rowspan="3">16.3</td></tr>
<tr><td>Jun</td><td>7.2</td><td>6.0</td></tr>
<tr><td>Jul</td><td>3.0</td><td>2.5</td></tr>
<tr><td>Aug</td><td>2.5</td><td>2.1</td><td rowspan="5">30.2</td><td rowspan="5">35.0</td></tr>
<tr><td>Sep</td><td>1.7</td><td>1.4</td><td rowspan="4">26.1</td><td rowspan="4">31.5</td></tr>
<tr><td>Oct</td><td>3.3</td><td>3.1</td></tr>
<tr><td>Nov</td><td>16.2</td><td>18.7</td></tr>
<tr><td>Dec</td><td>6.5</td><td>9.7</td></tr>
</table>

Month	Monthly		Seasonal total			
			Migration free		Full breeding	
	Scenario a	Scenario b	Scenario a	Scenario b	Scenario a	Scenario b
Total	93.1	**115.4**	**93.1**	**115.4**		**115.5**

70. There are several aspects of the trends in these data which argue against undertaking an assessment as precautionary as that proposed by Natural England. Of the two sites (East and West), the higher density of kittiwakes, and thus higher annual collision risks, were recorded on Norfolk Vanguard East and therefore scenario (b) with a higher proportion of turbines in this site (50%) represents the worst case for collisions overall. However, this site is almost 30 km further away from FFC SPA (minimum distance 235km) than Norfolk Vanguard West (minimum distance 205km), and therefore it would be expected that the abundance of kittiwakes in the breeding season would be higher on Norfolk Vanguard West. This observation is thus at odds with the suggested levels of connectivity (the opposite pattern would be expected). Furthermore, rather than increasing numbers being recorded in the wind farm sites as the breeding season progresses, as has been suggested would be the case on the basis of the tracking observations (Wischnewsi et al. 2018), what was actually observed was a negative trend in density between April and August with very low densities in June to August (0.04 birds/km^2) on Norfolk Vanguard East. Densities were also higher on Norfolk Vanguard East in the early months of the full breeding season (March and April) which are those also identified as migration months in Furness (2015). Furness (2015) states that:

Peak spring migration occurs in January-April in Belgium (Vanermen et al. 2013), in March-April generally in Europe (Cramp et al. 1977-94; Forrester et al. 2007). Peak numbers observed in spring at Trektellen seawatching UK sites (predominantly in south and east England) occurred in March.

71. Taken together, these observations have a poor correspondence with the suggestion that breeding adults from FFC SPA make up the majority (if not all) of the kittiwakes present on Norfolk Vanguard.

72. Thus, given the locations of Norfolk Vanguard East and West and the pattern of observations across the two sites, including March and April as breeding months for FFC SPA birds, this almost certainly over-estimates the number of collisions assigned to this population since there will be large numbers of migrants still passing through at this time. Across the two years, the surveys in March were conducted around the middle of March (12-14th) and early to middle April (4th and 5th and 13th and 15th). These dates are clearly consistent with the migration period (i.e. not conducted at the ends of the period) and further highlight the high degree of precaution in the

request from Natural England that FFC SPA birds should be considered to be the only birds present between March and August (i.e. 100% of collisions in those months should be assigned to the SPA).

73. As noted above, the FFC SPA apportioning estimate of 26.1% calculated above is considered precautionary, since it only incorporates UK immature birds and does not include consideration of the potentially very large number of birds from the Russian and Norwegian populations, of which an unknown, but likely very large, proportion will be present in the North Sea during migration and the breeding season.

74. Taking all these aspects together, the estimated seasonally apportioned collision estimates are provided in Table 10.

Table 10. Kittiwake seasonal and annual collision risk after application of apportioning rates (7.2% in spring, 26.1% in breeding and 5.4% in autumn) to the Flamborough and Filey Coast SPA using the migration free (May to July) and full (March to August) breeding seasons.

Scenario	Breeding season	Spring	Breeding	Autumn	Annual
A (67:33)	Migration-free	3.4	4.2	1.6	9.2
	Full	2.0	10.4	1.4	13.8
B (50:50)	Migration-free	4.6	4.3	1.9	10.8
	Full	2.8	11.8	1.7	16.3

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015).

75. As discussed above, apportioning breeding season mortality at the more distant Norfolk Vanguard East to the SPA is considered highly precautionary. Indeed, the tracking evidence provides very little evidence for connectivity to Norfolk Vanguard East at all. However, there is some evidence for connectivity with the slightly closer Norfolk Vanguard West site, although even here the migration free season is considered more appropriate for assigning collisions to the SPA. Nonetheless, consideration for the full breeding season for Norfolk Vanguard West is also presented.

76. Therefore, the Norfolk Vanguard FFC collision prediction comprises:

- Breeding season collisions at Norfolk Vanguard West, multiplied by the apportioning rate of 26.1%; and,
- Spring and autumn collisions at both Norfolk Vanguard East and West using the higher estimates from scenario (b) at apportioning rates of 7.2% and 5.4% respectively.

77. Thus, using the total breeding season collision estimate at Norfolk Vanguard West of 19.5 (for 120 10MW turbines, as per scenario (a)), gives an FFC (full) breeding season estimate (at 26.1%) of 5.1 individuals. This has been combined with the apportioned

spring and autumn migration estimates for scenario (b), the worst case scenario, of 2.8 and 1.7 to give a total FFC SPA mortality at Norfolk Vanguard of 9.6. The equivalent migration free estimate is 10.2 (=14.3 x 0.261 breeding, plus 64.1 x 0.072 spring, plus 35.0 x 0.054 autumn). Summing the upper 95% confidence intervals for these seasonal estimates (9.8, 3.8 and 2.3 respectively) the FFC annual total is 15.9, while the lower 95% confidence estimate is 4.3.

78. Although FFC SPA is much the largest kittiwake breeding colony in the southern North Sea, there are other, closer kittiwake colonies to Norfolk Vanguard West. The most recent population estimates for these have been extracted from the JNCC Seabird Monitoring Programme website (http://jncc.defra.gov.uk/smp/). These have been used to calculate the relative proportions from each colony which could be present on Norfolk Vanguard West (Table 11). It is important to note that this only provides an estimate of the relative proportions of breeding adults within that at sea population, and not the proportion of all birds present (i.e. including immature birds).

Table 11. Colonies of kittiwake between Humberside and Suffolk and estimated proportions of adults from each colony present on the Norfolk Vanguard site based (calculated using SNH tool[1]).

Colony	Minimum distance from Norfolk Vanguard West (km)	Approximate no. of breeding pairs (year)	Colony weighting (population size / distance^{2})	Colony proportion (colony weight / ∑ colony weights)
FFC SPA	205	45,504 (2017)	1.083	0.864
Lowestoft	57	325 (2016)	0.100	0.079
Sizewell	85	502 (2008)	0.069	0.055

79. The apportioning indicates that of the adults present, up to 86% are potentially from FFC SPA. On this basis, 22.6% of the total birds on the wind farm (86% of 26.1%) could originate from FFC in the breeding season. This is further evidence that the value of 26.1% (as calculated above) is precautionary.

80. Therefore, in summary:

a. There is very little evidence for connectivity between the FFC SPA and Norfolk Vanguard East site, with no tracking connectivity, and monthly trends in abundance which are much more compatible with migration movements than breeding movements. Therefore, since Norfolk Vanguard West is closer to FFC SPA and there is more compelling evidence for breeding season

[1] https://www.nature.scot/sites/default/files/2017-07/A2176850%20-%20Interim%20Guidance%20on%20Apportioning%20Impacts%20from%20Marine%20Renewable%20Developments%20to%20breeding%20seabird%20populations%20in%20special%20Protection%20Areas%20-%2021%20Dec%202016.pdf

connectivity on this site the HRA for Norfolk Vanguard combines the breeding seasons collisions at this site with the combined collisions across both sites in the migration seasons.

b. Since monthly patterns of abundance (on both sites) more closely correspond to migration movements, the migration free breeding season is considered more appropriate (although the full season is also presented); and

c. The proportion of the birds on Norfolk Vanguard West in the breeding season predicted to originate from the FFC SPA has been calculated using a precautionary rate of 26.1%. This is precautionary because it does not allow for the presence of breeding adults from closer colonies, nor that of Russian and Norwegian immatures.

81. The Norfolk Vanguard annual collisions apportioned to the FFC SPA using the full breeding season is 9.6. This combines several sources of precaution:

- Use of a nocturnal activity rate of 50% (Furness et al. in prep. Indicates that a value less than 20% is more appropriate for this species); and
- Bowgen and Cook (2018) recently estimated a kittiwake collision avoidance rate from an empirical study of 99%, which would reduce collisions by around 10% compared with the current predictions using 98.9%.

82. The SPA population at designation was 44,520 pairs (89,040 individuals). At an average natural adult mortality rate of 0.146, the natural annual mortality of the population is 13,000. The addition of up to 9.6 individuals would therefore increase the mortality rate by 0.07% (0.12% using the upper 95% confidence interval and 0.03% using the lower 95% confidence interval). Increases in mortality of less than 1% are considered to be undetectable against natural variation and therefore, the conclusion is that there will be no adverse effect on the integrity of the Flamborough and Filey Coast SPA as a result of kittiwake collisions at the proposed Norfolk Vanguard project.

3.2.1.3 EIA cumulative and HRA In-combination

83. Natural England advised that the in-combination collision assessment should include estimates for three additional Scottish wind farms (Hywind, Kincardine and Moray West) and that there is uncertainty regarding the appropriate values to use for the Hornsea Project Three and Thanet Extension wind farms as these are also currently in examination and therefore there is potential for variation. Following the Applicant's understanding from discussions with Natural England, estimates for Hornsea Project Three have been taken from that project's ES and for Thanet Extension from that project's submission at Deadline 3 (Vattenfall 2019b). Natural England also advised that for other wind farms with potential connectivity to the FFC

SPA during the breeding season, the apportioning rates presented for the East Anglia THREE wind farm, labelled as 'NE Method' should be used. These were: 100% for Lincs, Humber Gateway, Westermost Rough, Dudgeon, Race Bank and Triton Knoll; 83% for Hornsea Projects One and Two (NB, for Project One this was given as 66.6%, but NE advises that the higher rate for Project Two should be used) and 19.3% for the Dogger Bank Projects. In addition, for Hornsea Project Three a value of 94% was advised. These advised percentages have been used together with the value of 26.1% estimated for Norfolk Vanguard. As set out above, in accordance with Natural England's advice, cumulative totals without Hornsea Project THREE are also provided. Table 12 presents the full cumulative and in-combination predictions.

Table 12. Kittiwake collision mortality for all wind farms, and collisions apportioned to the Flamborough and Filey Coast SPA

Wind farm	Spring		Breeding		Autumn		Annual	
	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA	Total	FFC SPA
Beatrice Demonstrator	1.7	0.1	0.0	0.0	2.1	0.1	3.8	0.2
Greater Gabbard	11.4	0.8	1.1	0.0	15.0	0.8	27.5	1.6
Gunfleet Sands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Kentish Flats	0.7	0.1	0.0	0.0	0.9	0.0	1.6	0.1
Lincs	0.7	0.0	0.7	0.7	1.2	0.1	2.6	0.8
London Array	1.8	0.1	1.4	0.0	2.3	0.1	5.5	0.3
Lynn and Inner Dowsing	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scroby Sands	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sheringham Shoal	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Teesside	2.5	0.2	38.4	0.0	24.0	1.3	64.9	1.5
Thanet	0.4	0.0	0.3	0.0	0.5	0.0	1.2	0.1
Humber Gateway	1.9	0.1	1.9	1.9	3.2	0.2	7.0	2.2
Westermost Rough	0.1	0.0	0.1	0.1	0.2	0.0	0.5	0.1
Hywind	0.9	0.1	16.6	0.0	0.9	0.0	18.3	0.1
Kincardine	1.0	0.1	22.0	0.0	9.0	0.5	32.0	0.6
Beatrice	39.8	2.9	94.7	0.0	10.7	0.6	145.2	3.4
Dudgeon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Galloper	31.8	2.3	6.3	0.0	27.8	1.5	65.9	3.8
Race Bank	5.6	0.4	1.9	1.9	23.9	1.3	31.4	3.6
Rampion	29.7	2.1	54.4	0.0	37.4	2.0	121.5	4.2
Hornsea Project One	20.9	1.5	44.0	36.5	55.9	3.0	120.8	41.0
Blyth Demonstration Project	1.4	0.1	1.4	0.0	2.3	0.1	5.1	0.2
Dogger Bank Creyke Beck Projects A and B	295.0	21.2	288.0	55.6	135.0	7.3	718.0	84.1
East Anglia ONE	46.7	3.4	1.5	0.0	161.0	8.7	209.2	12.1
European Offshore Wind Deployment Centre	1.1	0.1	11.8	0.0	5.8	0.3	18.7	0.4
Firth of Forth Alpha and Bravo	247.6	17.8	153.1	0.0	313.1	16.9	713.8	34.7
Inch Cape	63.5	4.6	13.1	0.0	224.8	12.1	301.4	16.7

Wind farm	Spring Total	Spring FFC SPA	Breeding Total	Breeding FFC SPA	Autumn Total	Autumn FFC SPA	Annual Total	Annual FFC SPA
Moray Firth (EDA)	19.3	1.4	43.6	0.0	2.0	0.1	64.9	1.5
Neart na Gaoithe	4.4	0.3	32.9	0.0	56.1	3.0	93.4	3.3
Dogger Bank Teesside Projects A and B	216.9	15.6	136.9	26.4	90.7	4.9	444.5	46.9
Triton Knoll	45.4	3.3	24.6	24.6	139.0	7.5	209.0	35.4
Hornsea Project Two	3.0	0.2	16.0	13.3	9.0	0.5	28.0	14.0
East Anglia THREE	37.6	2.7	6.1	0.0	69.0	3.7	112.7	6.4
Hornsea Project Three	11.4	0.8	165.3	153.7	61.3	3.3	238.0	157.9
Thanet Extension	15.3	1.1	2.3	0.0	5.3	0.3	23.0	1.4
Moray West	7.0	0.5	79.0	0.0	24.0	1.3	110.0	1.8
Norfolk Vanguard	38.7	2.8	45.3	5.1	31.5	1.7	115.4	9.6
Total (inc. Hornsea Project Three)	**1205.2**	**86.7**	**1304.7**	**319.8**	**1544.9**	**83.2**	**4054.8**	**490**
Total (exc. Hornsea Project Three)	**1193.8**	**85.9**	**1139.4**	**166.1**	**1483.6**	**79.9**	**3816.8**	**332.1**

3.2.1.3.1 Cumulative assessment

84. The cumulative total, all age class annual kittiwake collision mortality is estimated to be 4,054.8 with the inclusion of Hornsea Project Three and 3,816.8 without this project. Note, however that many of the collision estimates for other wind farms were calculated on the basis of consented designs with higher total rotor swept areas than have been installed (or are planned), which is a key factor in collision risk. For example, the Beatrice wind farm, which is currently under construction, was consented on the basis of up to 125 x 7MW turbines but only 84 (of the same model) will be installed, leading to a reduction in mortality risk of 33%. A method for updating collision estimates for changes in wind farm design was presented in Trinder (2017). Updating the collision estimates for the Beatrice wind farm using this approach reduces the predicted annual mortality from 145 to 97. Applying the same method to the other wind farms in Table 12 can achieve a reduction in the cumulative annual mortality of around 550. Therefore, the values presented in Table 12, as well as being based on precautionary calculation methods, can be seen to overestimate the total risk by around 14% due to the reduced collision risks for projects which undergo design revisions post consent.

85. A review of nocturnal activity in kittiwakes (Furness, in prep.) has found that the value previously used for this parameter (50%) to estimate flight activity at night is a considerable overestimate and has identified evidence-based rates of 20% during the breeding season and 17% during the nonbreeding season.

86. It is straightforward to adjust mortality estimates using the new and old nocturnal activity rates and the monthly number of daytime and night time hours (i.e. it is not necessary to rerun the collision model for this update). However, it is necessary to calculate a mortality adjustment rate for each month at each wind farm because the duration of night varies with month and latitude (both of which are inputs to the collision model). This has not been undertaken for the current assessment but would be expected to reduce the cumulative total by at least 10%. This further emphasises the precautionary nature of the current assessment.

87. For the assessment of the adjacent East Anglia THREE wind farm a kittiwake population model was developed to assess the potential effects of cumulative mortality on the kittiwake BDMPS populations (EATL, 2015). Both density independent and density dependent models were developed. The outputs were presented in relation to additional adult mortality, therefore the cumulative total estimate here has been multiplied by 0.53 (Furness 2015) to estimate the adult component of the cumulative total, giving a figure of 2,149. For annual mortality of 2,250 (the nearest modelled mortality), the density dependent model predicted the population after 25 years would be 2.2% to 3.0% smaller than that predicted in the absence of additional mortality, while the more precautionary density independent model predicted equivalent declines of 6.8% to 7.1%. The population growth rate reduction for this level of mortality was estimated to be 0.3% using the density independent model and <0.01% using the density dependent model.

88. To place these predicted magnitudes of change in context, over three approximate 15 year periods (between censuses) the British kittiwake population changed by +24% (1969 to 1985), -25% (1985 to 1998) and -61% (2000 to 2013) (http://jncc.defra.gov.uk/page-3201 accessed 26th August 2015). Changes of between 2% and 7% across a longer (modelled 25 year) period against a background of natural changes an order of magnitude larger will almost certainly be undetectable. Although the Norfolk Vanguard application is for a 30 year project life time, compared with the above PVA span of 25 years, the additional 5 years will make very little difference to the growth rate predictions from either the density dependent or density independent models.

89. Natural England advised that the results from density independent models should be used ‘where there is no information on population regulation for the focal population’ (NE 2017).

90. Evidence for density dependent regulation of the North Sea kittiwake population was summarised in EATL (2016b). While Natural England accepted there was strong evidence for the presence of density dependence operating in the population they maintained that because its mode of operation was less clear the results of the

density independent PVA models should be used in preference to the density dependent ones (acknowledging that using these results the predictions were the most precautionary ones). However, Trinder (2014) explored a range of strengths of density dependence for this species and identified model parameters which produced population predictions consistent with patterns of seabird population growth which have been observed across a wide range of taxa (inc. kittiwake) worldwide (Cury et al. 2011). Thus, there is robust evidence for density dependent regulation of the North Sea kittiwake population (and for seabirds more widely) and its inclusion in the kittiwake population model (EATL 2015) balanced this evidence with reasonable precaution. Consequently, the density dependent kittiwake model results are considered to be the more robust ones on which to base this assessment.

91. Kittiwake is considered to be of low to medium sensitivity, low to medium conservation value and the magnitude of effect described above is considered to be low. Consequently, the worst case cumulative collision mortality is considered to be of low magnitude, resulting in impacts of **minor adverse** significance. However, when the various sources of precaution are taken into account (precautionary avoidance rate estimates, reduction in wind farm sizes, over-estimated nocturnal activity) the cumulative collision risk impact magnitude is almost certainly smaller still.

3.2.1.3.2 In-combination assessment

92. The in-combination adult total annual kittiwake collision estimate is 490, of which Norfolk Vanguard contributes 9.6 (1.9%), although it should be noted that this is considered to be an over-estimate due to the precautionary assumptions noted above. Without Hornsea Project Three this total is 332.1 (of which Norfolk Vanguard contributes 2.9%).

93. The increase in the background mortality due to this in-combination collision risk is 3.8% with the inclusion of Hornsea Project Three, and 2.5% without Hornsea Project Three.

94. A population model was produced for this population for the Hornsea Project Three wind farm (MacArthur Green 2018). This model was an update of similar models produced for Hornsea Project Two, with the addition of a matched-run approach for calculating counterfactual outputs and an extended simulation period (35 years). Simulations were conducted with and without density dependence and were summarised as the counterfactual of population size and population growth rate. The outputs from these models for adult mortality levels of 350 and 500 (the closest upper values to these totals) are provided in Table 13.

Table 13. Kittiwake FFC SPA population modelling results from MacArthur Green (2018).

Model	Mortality	Counterfactual metric (after 30 years)		Source table (MacArthur Green 2018)
		Growth rate	Population size	
Rate set 1, density independent	350	0.996	0.892	Table A2 5.1 & 5.3
	500	0.994	0.849	
Rate set 1, density dependent	350	0.999	0.968	Table A2 6.1 & 6.3
	500	0.999	0.954	
Rate set 2, density independent	350	0.996	0.892	Table A2 7.1 & 7.3
	500	0.994	0.850	
Rate set 2, density dependent	350	0.999	0.966	Table A2 8.1 & 8.3
	500	0.999	0.946	

95. The maximum reduction in the population growth rate, at a mortality of 500, using the more precautionary density independent model was 0.6% (0.9947) and without Hornsea Project Three this was 0.4%. Using the more realistic density dependent model these maximum reductions in growth rate were 0.1% (0.999) both with and without Hornsea Project Three.

96. This growth rate reduction represents a very small risk to the population's conservation status.

97. The kittiwake breeding numbers at the Flamborough and Filey Coast SPA have remained relatively stable around an average of approximately 40,000 pairs over the last 20 years. The RSPB reported that since 2000 the population has grown by 7% which would equate to 0.4% annual growth rate (RSPB unpublished report). Therefore, the kittiwake population appears to be in favourable conservation status and the relevant conservation objective is to maintain this status, subject to natural change. On the basis of the precautionary in-combination collision estimate (including over-estimates for consented vs. built designs and over-estimated nocturnal activity) combined with the precautionary density independent model predictions for the total adult mortality of 490, there may be to be a small risk that further population growth will be restricted. However, the much more realistic density dependent model suggests that this level of mortality will have a much smaller effect on the population, with only a very slight reduction in the growth rate, and that the population's conservation status will not be affected.

98. Natural England contends that density dependence should only be included in population models when evidence for this is available for the population in question and that this is not the case for the Flamborough and Filey Coast SPA kittiwake population. However, as noted above, there is evidence for density dependence in

the North Sea kittiwake population (EATL 2016) and exploratory analysis has been used to guide the most appropriate method for inclusion in population models (Trinder 2014). Therefore, while there may not be direct evidence for the SPA population, there is evidence of density dependence for the wider population of which it is an integral part and there is no reason that the SPA population would not be affected by the same regulatory drivers. Therefore, the arguments against the inclusion of density dependence are not considered to apply in this case.

99. Therefore, the conclusions presented in the Norfolk Vanguard ES and HRA and subsequent submissions (ExA; AS; 10.D6.17) remain the same; it can be concluded that there will be no adverse effect on the integrity of Flamborough & Filey Coast SPA from collision impacts on kittiwake due to the proposed Norfolk Vanguard project in-combination with other plans and projects.

100. Furthermore, the contribution from Norfolk Vanguard to this total has been substantially reduced following design revisions to mitigate collision risks, with the annual collision mortality estimate for kittiwake reduced by 67% when the removal of the 9MW turbine, revised layout and turbine draught height increase are considered together.

3.3 Herring gull

3.3.1 Collision risk

3.3.1.1 EIA Project alone

101. The revised collision risk estimates for herring gull for the 10MW turbine and the revised project layout (ExA; CRM 10.D6.5.1) and 5m turbine draught increase (ExA;AS;10.D.7.5.2), calculated using the Band (2012) deterministic model and Natural England's preferred parameter values, are provided in Table 14.

Table 14. Herring gull seasonal and annual collision risk using the migration free (April to August) and full (March to September) breeding seasons.

Breeding season	Migration free breeding	Midwinter/non-breeding	Annual
Migration-free	0.46 (0-1.62)	12.99 (7.24-21.13)	13.45 (7.24-22.75)
Full	0.76 (0-2.8)	12.7 (7.24-19.95)	

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015).

102. In the submission at Deadline 6.5 (ExA; CRM 10.D6.5.1, Table 2) it was concluded that a higher annual mortality of 17.9 (prior to the turbine draught height increase) would not increase the background rate by more than 1% and therefore it was concluded that Norfolk Vanguard alone would have no significant impact at the EIA scale. This conclusion is further supported by the lower revised annual estimate of

13.4 (a reduction of 25% for the turbine draught height increase alone), and therefore the conclusion of no significant impact at the EIA scale remains valid.

3.3.1.2 EIA Cumulative

103. Natural England requested the inclusion of a cumulative assessment of herring gull collision risk.

104. The cumulative herring gull collision risk prediction is presented in Table 15. This collates collision predictions from other wind farms which may contribute to the cumulative total. This table takes the wind farm assessment for East Anglia THREE as its starting point and adds more recent wind farm predictions.

105. The collision values presented in Table 15 include totals for breeding, nonbreeding and annual periods. However, not all projects provide a seasonal breakdown of collision impacts, therefore it is not possible to extract data from these periods for cumulative assessment. Natural England has previously noted that an 80:20 split between the nonbreeding and breeding seasons is appropriate for lesser black-backed gull in terms of collision estimates (Natural England, 2013), and this has been used for herring gull. Therefore, for those sites where a seasonal split was not presented the annual numbers in Table 15 have been multiplied by 0.8 to estimate the nonbreeding component and 0.2 to estimate the breeding component.

Table 15. Herring gull cumulative collision risk.

Wind farm	Breeding	Nonbreeding	Annual
Beatrice Demonstrator	0.0	0	0.0
Greater Gabbard	0.0	0	0.0
Gunfleet Sands	0.0	0	0.0
Kentish Flats	0.5	1.7	2.2
Lincs	0.0	0	0.0
London Array	0.0	0	0.0
Lynn and Inner Dowsing	0.0	0	0.0
Scroby Sands	0.0	0	0.0
Sheringham Shoal	0.0	0	0.0
Teesside	8.7	34.5	43.2
Thanet	4.9	19.6	24.5
Humber Gateway	0.4	1.1	1.5
Westermost Rough	0.1	0.0	0.1
Hywind	0.6	7.8	8.4
Kincardine	1.0	0.0	1.0
Beatrice	49.4	197.4	246.8
Dudgeon	0.0	0	0.0
Galloper	27.2	0	27.2
Race Bank	0.0	0	0.0
Rampion	155.0	0	155.0
Hornsea Project One	2.9	11.6	14.5

Wind farm	Breeding	Nonbreeding	Annual
Blyth Demonstration Project	0.5	2.2	2.7
Dogger Bank Creyke Beck Projects A and B	0.0	0	0.0
East Anglia ONE	0.0	28.0	28.0
European Offshore Wind Deployment Centre	4.8	0	4.8
Firth of Forth Alpha and Bravo	10.0	21.0	31.0
Inch Cape	0.0	13.5	13.5
Moray Firth (EDA)	52.0	0	52.0
Neart na Gaoithe	5.0	12.5	17.5
Dogger Bank Teesside Projects A and B	0.0	0	0.0
Triton Knoll	0.0	0	0.0
Hornsea Project Two	23.8	0	23.8
East Anglia THREE	0.0	23.0	23.0
Hornsea Project Three	1.0	7.0	8.0
Thanet Extension	10.0	4.0	14.0
Moray West	12.0	1.0	13.0
Norfolk Vanguard	0.8	12.7	13.5
Total (inc. Hornsea Project Three)	**370.6**	**398.6**	**769.2**
Total (exc. Hornsea Project Three)	**369.6**	**391.6**	**761.2**

3.3.1.2.1 Cumulative assessment

106. On the basis of the worst case Norfolk Vanguard collision estimate the annual cumulative total including Hornsea Project Three is 769.2 and without this project is 761.2.

107. The background mortality for the largest BDMPS population (466,511) at an all age class average mortality rate of 0.174 (Appendix 3.2, document reference ExA; WQApp3.2; 10.D1.3) is 81,173. The addition of 769.2 to this increases the rate by 0.95%, and without Hornsea Project Three this would be 0.94%. These are below the 1% threshold of detectability.

108. This total also includes, among other sources of precaution, over-estimated nocturnal activity for projects and the use of consented collision estimates for projects which have since been constructed to designs with much lower collision risks.

109. Nonetheless, even including these additional sources of precaution the cumulative herring gull collision risk results in an impact of minor magnitude and a **minor adverse** significant impact.

3.4 Lesser black-backed gull

3.4.1 Collision risk

3.4.1.1 EIA Project alone

110. The revised collision risk estimates for lesser black-backed gull for the 10MW turbine and the revised project layout (ExA; CRM 10.D6.5.1) and 5m turbine draught height increase (ExA;AS;10.D7.5.2), calculated using the Band (2012) deterministic model and Natural England's preferred parameter values, are provided in Table 16.

Table 16. Lesser black-backed gull seasonal and annual collision risk using the migration free (May to July) and full (April to August) breeding seasons.

Breeding season	Spring	Migration free breeding	Autumn	Midwinter / nonbreeding	Annual
Migration-free	1.23 (0-4.38)	7.25 (2.15-14.52)	12.9 (4.36-24.43)	1.67 (0-4.06)	23.05 (6.51-47.38)
Full	0.56 (0-2.23)	15.57 (4.97-30.44)	5.25 (1.53-10.65)	1.67 (0-4.06)	

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015).

111. In the submission at Deadline 6.5 (ExA; CRM 10.D6.5.1, Table 2) it was concluded that a higher annual mortality of 31.7 (prior to the turbine draught height increase) would not increase the background rate by more than 1% and therefore it was concluded that Norfolk Vanguard alone would have no significant impact at the EIA scale. This conclusion is further supported by the lower revised annual estimate of 23.1 (a reduction of 27% for the turbine draught height increase alone), and therefore the conclusion of no significant impact at the EIA scale remains valid.

3.4.1.2 Apportioning to the Alde Ore Estuary SPA

112. Alde-Ore Estuary SPA is located 92 km from the closest point of the Norfolk Vanguard OWF sites. The lesser black-backed gull is estimated to have a mean breeding season foraging range of 72 km from colonies, a mean maximum foraging range of 141 km, and a maximum recorded foraging range of 181 km (Thaxter et al. 2012). Therefore, breeding adults from Alde-Ore Estuary SPA may forage over an area that includes the Norfolk Vanguard site, although the site is further from the colony than most likely foraging activity of this population. Other breeding lesser black-backed gull SPAs in Britain are located more than 181km from the Norfolk Vanguard site. The Alde-Ore Estuary SPA is therefore the only British lesser black-backed gull SPA colony that is within maximum foraging range.

113. Natural England advised the Applicant that consideration should be given to presentation of a range of percentages for the proportion of birds on the Norfolk Vanguard site which may originate from this SPA, with an upper limit of 30% (Natural

England Deadline 7 submission, EN010079 280590 Norfolk Vanguard Natural England's Comments by species on Vanguard Deadline 6 (REP6-021) and Deadline 6.5 (AS-043) information). The following sections present a detailed review of the evidence relating to lesser black-backed gull behaviour, foraging ecology and the regional population, in order to arrive at appropriate rates for this assessment. A key aspect of this review was the need to identify an appropriate balance between uncertainty and precaution.

114. As well as the Alde-Ore Estuary SPA, there are non-SPA colonies of lesser black-backed gulls located within foraging range of Norfolk Vanguard, including rooftop nesting gulls in several towns in Suffolk and Norfolk. As there is a high likelihood that birds from these populations will also be present on Norfolk Vanguard it is appropriate to consider the relative population sizes and potential for connectivity. This is discussed in detail below.

115. The national census of seabirds breeding in Britain and Ireland in 1985-86 found 37 pairs of lesser black-backed gulls breeding in Norfolk and fewer than 43 pairs in Suffolk at sites outside the Alde-Ore Estuary SPA (Lloyd et al. 1991). There were at least 5,000 pairs nesting at Orfordness in the Alde-Ore Estuary SPA and 2 or 3 pairs at Havergate (Lloyd et al. 1991 and JNCC Seabird Monitoring Programme (SMP) database), so the Alde-Ore Estuary SPA held 98% of the lesser black-backed gulls breeding in East Anglia in 1985-86. The national census of seabirds breeding in Britain and Ireland in 1998-2002 found 1,605 pairs of lesser black-backed gulls breeding in Norfolk and 1,166 pairs in Suffolk at sites outside the Alde-Ore Estuary SPA (Mitchell et al. 2004), so 2,771 pairs were found nesting at sites in East Anglia away from the Alde-Ore Estuary SPA. The JNCC SCM (Site Condition Monitoring) database shows a huge drop in breeding numbers at Orfordness and Havergate at that time after many years of colony growth (Plate 2.1). According to JNCC, this was apparently caused by foxes which were entering the colony to kill adults and chicks and take gull eggs (Mavor et al. 2001). Numbers have declined further since 2001 (Plate 2.1), as the problem of depredations by foxes has apparently continued.

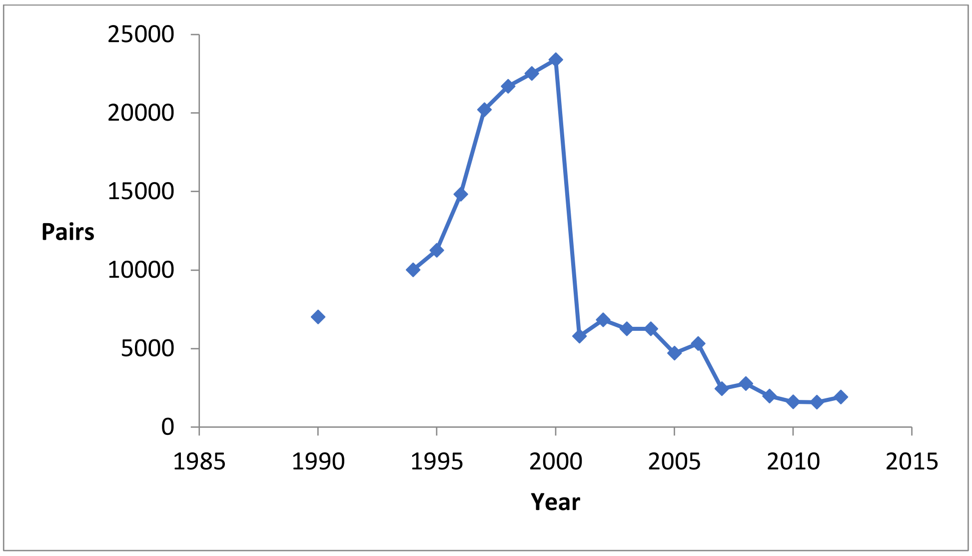

Plate 3.1 Number of breeding pairs of lesser black-backed gulls in the Alde-Ore Estuary SPA; Orfordness plus Havergate (data from JNCC SCM database).

116. There were estimated to be 23,000 pairs at Orfordness and 400 pairs at Havergate in 2000, so an estimated 89% of the lesser black-backed gulls breeding in Norfolk and Suffolk were in the Alde-Ore Estuary SPA in 2000. The colony at Orfordness held 5,500 pairs, and the colony at Havergate held 290 pairs in 2001 (JNCC SMP database). That means that 68% of the breeding population was within the Alde-Ore Estuary SPA in 2001.

117. The Alde-Ore population of lesser black-backed gulls has since decreased considerably, the most recent published counts being 640 pairs at Orfordness in 2012 and 1,668 pairs at Havergate in 2016. It is unclear why no counts have been entered into the JNCC SMP database for Orfordness since 2012 and that limits understanding of any changes that have occurred since 2012.

118. By comparison, numbers breeding elsewhere in East Anglia have increased. There were 743 pairs at urban colonies in Great Yarmouth in 2012, 467 pairs at Southtown/Gorleston in 2012, probably about 2,000-3,000 pairs at Lowestoft in 2008-2011, and a few hundred pairs at other sites in Norfolk and Suffolk (Piotrowski 2013). These urban colonies have only been censused a few times, and counts are not very accurate because many rooftops are impossible to view, so the numbers are likely to be underestimates (Ross et al. 2016). Furthermore, the 2012 census of urban breeding gulls in Suffolk was carried out after adverse conditions resulted in considerable breeding failure of many gulls (Piotrowski 2013) so is also likely to have underestimated numbers at urban sites. However, despite the relatively incomplete census data, it is clear that urban colonies have been growing very fast, as seen at Lowestoft (Plate 2.2), and Great Yarmouth (Plate 2.3).

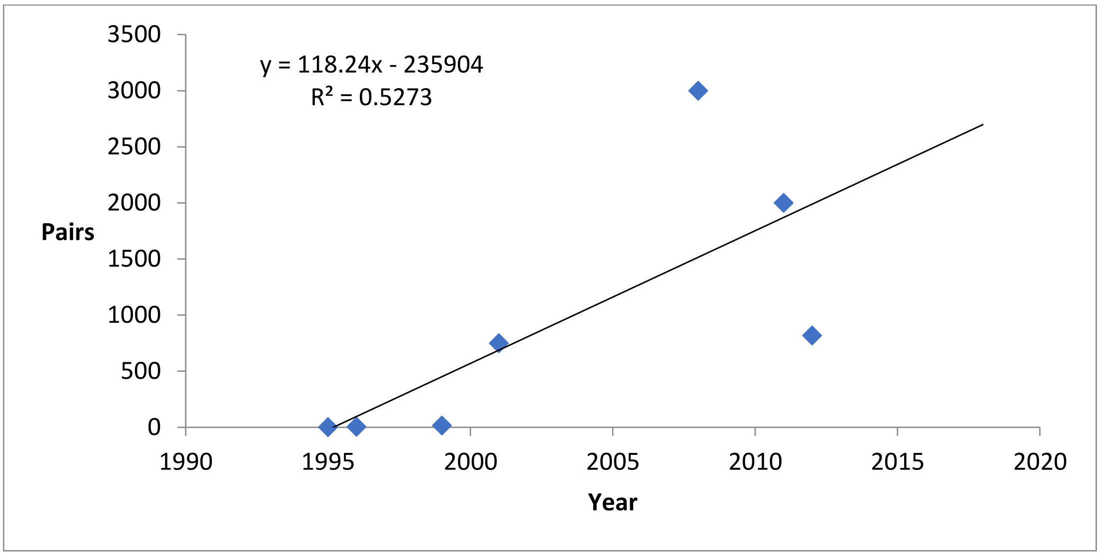

Plate 3.2 Number of breeding pairs of lesser black-backed gulls in Lowestoft (data from JNCC SCM database and Piotrowski 2013).

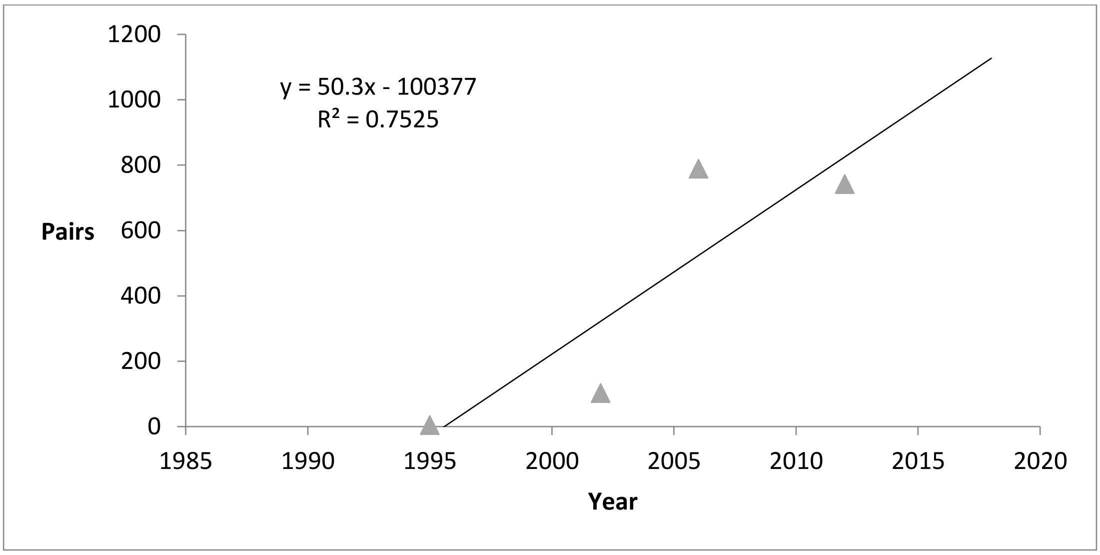

Plate 3.3 Number of breeding pairs of lesser black-backed gulls in Great Yarmouth (data from JNCC SCM database and Piotrowski 2013).

119. In addition, breeding numbers have increased at Felixstowe (1,401 pairs in 2013; Plate 2.4) and Ipswich (99 pairs in 2001, 262 pairs in 2012), which are also urban colonies, and remained relatively stable at Outer Trial Bank (1,704 pairs in 2006, 1,457 pairs in 2009 and 1,294 pairs in 2018) (JNCC SCM database).

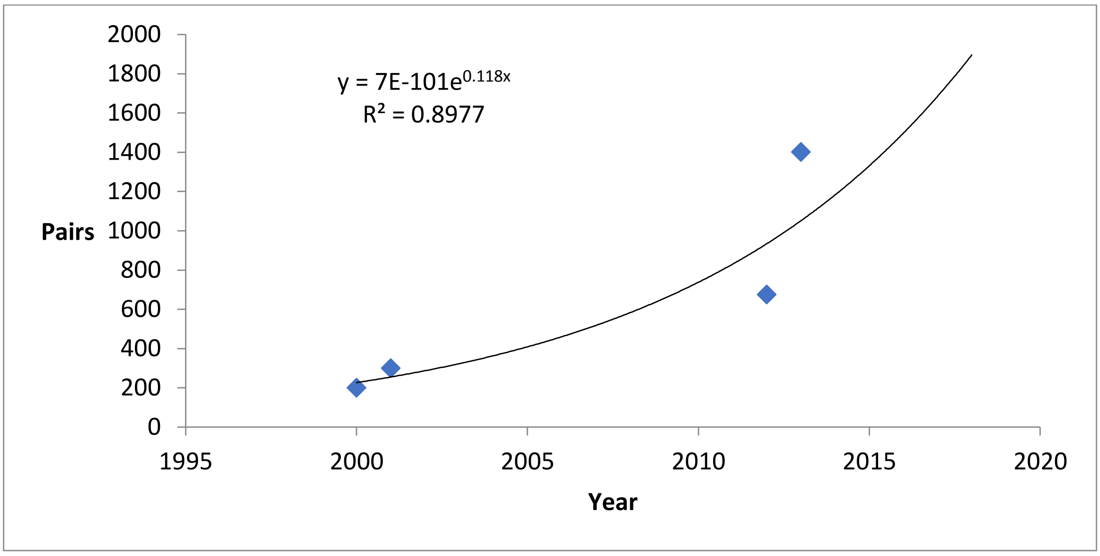

Plate 3.4 Numbers of breeding pairs of lesser black-backed gulls at Felixstowe (data from JNCC SCM database). For this colony an exponential growth curve is a better fit than a linear increase.

120. The numbers at Alde-Ore Estuary SPA colonies in 2012-2016 (ca. 2,300 pairs) compare with ca. 5,100 pairs at sites in Norfolk and Suffolk outside the SPA. This suggests that the percentage of Norfolk and Suffolk lesser black-backed gulls breeding within the SPA had fallen to about 31% of the population.

121. Concerted efforts to make urban areas 'gull-proof' can sometimes result in a reduction in breeding numbers of urban gulls of as much as 25% (Coulson and Coulson 2009) though such reductions may possibly only be temporary until gulls find other urban nest sites where they are tolerated. In general, urban nesting by gulls has increased throughout the UK much faster than total populations of gulls (Raven and Coulson 1997, Nager and O'Hanlon 2016) because the breeding success of gulls tends to be higher at urban sites than in rural colonies (chicks on rooftops are not exposed to predators such as foxes and are less at risk of disturbance or conflict with other gulls; Monaghan 1979, Monaghan and Coulson 1977), and survival of adults at urban colonies is at least as high, and probably higher, than at rural sites (Rock and Vaughan 2013, O'Hanlon and Nager 2018). Piotrowski (pers. comm. who carried out the census of breeding numbers at urban sites in Suffolk in 2012) stated that efforts to deter urban nesting gulls in Suffolk have largely been ineffective and do not seem to have resulted in significant reductions in the population in urban sites overall.

122. Urban nesting lesser black-backed gull numbers in Suffolk increased by over 1000% between 1995 and 2012 (Piotrowski 2013) at a period when numbers breeding in the Alde-Ore Estuary SPA decreased by about 70%. If this trend has continued then the proportion of lesser black-backed gulls at Norfolk Vanguard that originate from Alde-

Ore Estuary SPA may be decreasing further below 31% since 2012, but this is uncertain. At a qualitative level, the picture shown quantitatively in 2012 appears not to be much changed since then. However, a repeat census of breeding gull numbers would be helpful to check on that and may be carried out as part of the current national census of breeding seabirds and could be made more accurate by use of drones to photograph inaccessible rooftops (Ross et al. 2016, Rush et al. 2018).

123. The available data show that the Alde-Ore Estuary SPA held about 98% of the East Anglia breeding population of lesser black-backed gulls in 1985-86, 89% of the East Anglia breeding population of lesser black-backed gulls in 2000, 68% in 2001 and about 31% in 2012-2016 (Plate 2.5). Since numbers at urban colonies in particular have been on an upward trend, it seems likely that the percentage of the population within the Alde-Ore Estuary SPA will have decreased further since 2012-2016.

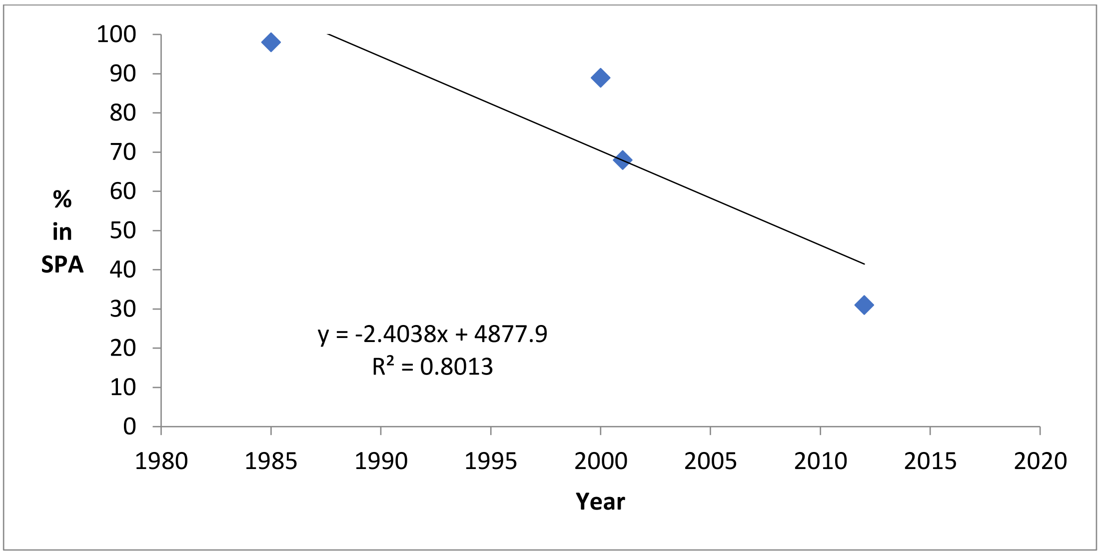

Plate 3.5 The percentage of lesser black-backed gulls breeding in East Anglia that were breeding within the Alde-Ore Estuary SPA in different survey years (based on JNCC SCM database and Piotrowski 2013).

124. It is likely that breeding adult lesser black-backed gulls visiting the Norfolk Vanguard site will tend to come from colonies within foraging range, and within that sample, may come more from colonies closer to the site than from colonies further away. In that context, it is worth noting that the SPA population at Alde-Ore Estuary is in the middle of the range of distances of East Anglian lesser black-backed gull colonies from Norfolk Vanguard (Table 17). Application of the simple population size – distance colony apportioning approach developed jointly by SNH (Scottish Natural Heritage) and MacArthur Green indicates that around 17% of the birds recorded on the Norfolk Vanguard site would be expected to originate from the Alde Ore Estuary SPA (Table 17).

Table 17. Colonies of lesser black-backed gulls in East Anglia ranked according to the minimum distance from Norfolk Vanguard.

Colony	Minimum distance from Norfolk Vanguard (km)	Approximate no. of breeding pairs in period 2008-2015	Colony weighting (population size / distance$^{2)}$	Colony proportion (colony weight / ∑ colony weights)
Great Yarmouth	51	750	0.288	0.21
Southtown	55	450	0.149	0.11
Lowestoft	60	2000	0.556	0.41
Alde-Ore Estuary SPA	92	2000	0.236	0.17
Felixstowe	120	700	0.049	0.04
Ipswich	120	250	0.017	0.01
Outer Trial Bank	140	1300	0.066	0.05

Noting the maximum foraging range of breeding lesser black-backed gulls is reported by Thaxter et al. (2012) as 181 km and estimated proportions of each present on the Norfolk Vanguard site based (calculated using SNH tool[2]).

125. On the basis of the population sizes and distances, of all the breeding adults present on Norfolk Vanguard in the breeding season, 17% are expected to be breeding adults from Alde Ore Estuary SPA. However, since adults comprise around 58% of the total population (Furness 2015), and since immature birds are more likely to visit areas distant from the main foraging areas, with locations close to colonies used by breeding adults (Wakefield et al. 2017), the overall proportion of birds at Norfolk Vanguard during the breeding season that are breeding adults is likely to be at most 58%, and possibly much less. Therefore, the proportion of birds at Norfolk Vanguard that are breeding adults from the Alde-Ore Estuary SPA is likely to be 17% of, at most, 58% of the total (i.e. approximately 10% overall). However, tracking data from adults breeding at the Alde-Ore Estuary SPA provide a better approach to estimating numbers at Norfolk Vanguard originating from that SPA and so tracking data are considered below.

126. It is likely that the amount of foraging within the marine environment varies among colonies and among years, depending on the relative availability of different feeding opportunities. Lesser black-backed gulls are generalist feeders, able to exploit a wide range of foods from urban waste food to earthworms on rural pasture land to small mammals and insects in grassland to intertidal animals, marine fish caught at sea and fisheries waste (discards and offal) made available behind fishing boats. However, there is evidence from diet studies and from tracking studies, that breeding adult lesser black-backed gulls tend to switch to feeding on marine fish

[2] https://www.nature.scot/sites/default/files/2017-07/A2176850%20-%20Interim%20Guidance%20on%20Apportioning%20Impacts%20from%20Marine%20Renewable%20Developments%20to%20breeding%20seabird%20populations%20in%20special%20Protection%20Areas%20-%2021%20Dec%202016.pdf

when rearing chicks. This is thought to be at least in part a strategy to provide chicks with nutritionally better food to support chick growth and development. That switch would, therefore, be just as appropriate for urban nesting gulls as for rural nesting gulls.

127. Tracking data (Hayley Douglas, pers. comm.) and diet data (Steve Piotrowski, pers. comm.) for urban nesting lesser black-backed gulls do indeed suggest that those birds feed to an extent in marine habitat, especially when rearing chicks, and do not suggest that urban nesting gulls are significantly less marine than those nesting in rural colonies (based on evidence reviewed below). Lesser black-backed gulls nesting in urban colonies in East Anglia include marine fish in their breeding season diet as well as earthworms, small mammals and urban food waste (Steve Piotrowski, pers. comm.). Those birds clearly forage at sea to some extent, just as some rural nesting gulls do.

128. Some rural nesting lesser black-backed gulls do not seem to feed at sea while breeding. Clewley et al. (2017) reported on tracking data from adult lesser black-backed gulls breeding at Bowland Fells SPA. Two individuals from this rural inland colony spent a small minority of their foraging time in the marine environment but less than 10 km from the coast, whereas 14 others were never tracked over marine habitat (although three spent a small amount of time in estuarine habitat). Scragg et al. (2016) tracked ten adult lesser black-backed gulls breeding at the Ribble and Alt Estuary SPA and found that even for this coastal population, over 90% of their position fixes away from the colony occurred inland, with less than 0.5% occurring in marine habitat. Those studies indicate that rural nesting lesser black-backed gulls can have very low connectivity with marine habitat, even when the colony is at the coast.

129. Tracking of urban nesting gulls has only begun very recently (Rock et al. 2016), is based on small sample sizes, and is mostly not yet published. The ‘tag-n-track’ project has deployed GPS tags on lesser black-backed gulls breeding on rooftops in Strathclyde (Scotland). The data show that different individuals tend to have particular individual habits (as often found in gulls; Navarro et al. 2017), often returning regularly to the same location. However, birds nesting on rooftops include individuals that forage in the Clyde Estuary and Clyde Sea (Hayley Douglas, pers. comm.). Tracking of a small sample of breeding lesser black-backed gulls nesting in Bristol indicates that those birds do not forage in marine habitat, presumably because the sea is too distant and there are adequate foraging opportunities within closer range (Anouk Spelt, pers. comm.). Coulson and Coulson (2008) found that lesser black-backed gulls nesting in Dumfries did not forage in marine habitat, but fed mainly on agricultural land, especially on earthworms. Thaxter et al. (2017)

estimated that up to 41 birds would need to be tracked for about 145 days in order to describe 95% of area use by the population. On that basis, no clear conclusions can be reached about the relative importance of marine versus terrestrial habitat use from tracking studies based on deployment of very few tags for short periods of time, but the studies mentioned above do indicate that some urban nesting lesser black-backed gulls will forage at sea, and also indicate that birds from some rural colonies will forage almost exclusively inland. There is no evidence that urban nesting lesser black-backed gulls show lower connectivity with marine foraging habitat than rural nesting lesser black-backed gulls, although that possibility cannot be ruled out.

130. Tracking data (Thaxter et al. 2015) indicate very low connectivity between breeding lesser black-backed gulls at Orfordness (Alde-Ore Estuary SPA) and the Norfolk Vanguard site. Connectivity appears to vary between zero and very low across the years studied, presumably depending on variations in food availability in different years. Tracking data show a time budget overlap with the former East Anglia Zone of 3.7% in 2010, 1.1% in 2011 and 0.2% in 2012 (Thaxter et al. 2015 Supplementary material Appendix A). The Norfolk Vanguard site forms a small part of the former East Anglia Zone. The tracking data indicate that much less than 0.5% of the foraging time of lesser black-backed gulls is spent within the Norfolk Vanguard site plus 2km buffer. For the population of about 2,000 breeding pairs at Alde-Ore Estuary SPA that would represent considerably fewer than 10 birds (0.5% of the total number of pairs) at any point in time (assuming that under normal circumstances one adult is at the nest site while the other is away on a foraging trip). Given that there were on average about 300 lesser black-backed gulls in the Norfolk Vanguard site during the breeding season (April to August), fewer than 10 birds during the chick-rearing period from the Alde-Ore would represent less than 3% of the lesser black-backed gulls present. This finding is consistent with the fact that the Alde-Ore Estuary SPA population (c. 2,000) represents only about 25% of the population of adult lesser black-backed gulls breeding in East Anglia (c. 7,500, although this total is likely to be incomplete and therefore an underestimate). It also corresponds with the observation that Norfolk Vanguard is located towards the upper limit of lesser black-backed foraging range from most breeding colonies and is therefore likely to be used more by nonbreeders than by breeding adults.

131. Tracking data are for chick-rearing periods, so do not necessarily apply at other times during the breeding season. However, lesser black-backed gulls show more marine foraging behaviour during chick-rearing and more terrestrial foraging behaviour earlier in the breeding season, so the overlap with Norfolk Vanguard is likely to be highest during the latter part of the breeding season when birds have chicks to provision and is probably lower than this during the early breeding season.

132. Given the low numbers indicated by tracking this raises the question of where birds observed on Norfolk Vanguard come from, if not Alde-Ore SPA. To be precautionary in relation to the SPA population of Alde-Ore Estuary, it has been assumed that no breeding adults from the populations in the Netherlands visit the Norfolk Vanguard site because tracking data from birds in the Netherlands strongly indicate that connectivity for these birds is extremely low (Camphuysen 1995, 2013; Camphuysen et al. 2015). However, it is known that there are large numbers of immature lesser black-backed gulls in the populations (Furness 2015 estimated from demographic data that about 40% of the population will be immature birds and 60% will be breeding age adults). While younger immature birds may remain in the wintering area year round, during spring and summer older immatures move towards breeding areas and may form a significant part of the population at sea in areas such as Norfolk Vanguard. Consequently, a substantial part of the birds present at Norfolk Vanguard is likely to be immature birds from a variety of populations drawn from a much larger area than just East Anglia. The birds present may also include breeding adults from non-SPA colonies in East Anglia, especially those closer to Norfolk Boreas than is the Alde-Ore Estuary SPA (such as Great Yarmouth, Southtown, and Lowestoft).

133. To conclude, during the breeding season, on the basis of relative population sizes and colony distance, combined with age ratios, the breeding adults from Alde-Ore Estuary SPA would comprise less than 17% of the on-site birds, while tracking data suggest this percentage would most likely be less than 3%. Both of these values have been used in the assessment for the breeding season and represent upper and lower limits on apportioning rates, derived from the available evidence.

134. During migration, lesser black-backed gulls of all age classes will pass through the southern North Sea, with a small proportion of these passing through the Norfolk Vanguard site. Therefore, during migration, birds from many different local populations within the region may be at risk of collision mortality and the Alde-Ore Estuary SPA population represents only a very small fraction of the regional population potentially at risk. The lesser black-backed gull Biologically Defined Minimum Population Scales (BDMPS) population in UK North Sea and Channel waters in autumn (August-October) is estimated to be 209,000 birds, while the spring (March-April) population is estimated to be 197,000 birds (Furness 2015). The total Alde-Ore SPA lesser black-backed gull population has been estimated at around 6,700 individuals (assuming adults comprise 60% of the population, Furness 2015). This indicates that birds associated with the Alde-Ore SPA represent about 3.3% of these BDMPS populations. Therefore, it is likely that about 3.3% of the estimated collision mortality during the autumn and spring migration periods would affect birds associated with the Alde-Ore SPA population, of which around 60% would be

breeding adults (i.e. 2% of the total collision mortality would be breeding adults from Alde-Ore Estuary SPA). This percentage applies both for estimated mortality due to the proposed Norfolk Vanguard project alone, and to in-combination effects within the region.

135. During winter, lesser black-backed gulls are present in UK waters in smaller numbers than during migration; the estimated BDMPS winter population of lesser black-backed gulls in the UK North Sea and Channel waters is about 39,000 birds (Furness 2015). Adults from the Alde-Ore SPA lesser black-backed gull breeding population may represent a higher proportion of the winter BDMPS than they do during the migration seasons BDMPS populations because a higher proportion of the overwintering birds are likely to be adults (most immatures migrate further south). Furness (2015) considered that around 50% of breeding adults from the SPA remain in the region (a precautionary assumption), hence the proportion of birds from the Alde-Ore SPA will be approximately 5% (Furness 2015). Hence, no more than 5% of the estimated collision mortality on the lesser black-backed gull population during winter would be apportioned to the Alde-Ore SPA breeding population, either for estimated mortality due to the proposed Norfolk Vanguard project alone, or in-combination for the region. The true percentage is an unknown amount below 5%, but is likely to be greater than the 3.3% estimated during migration seasons. Thus, a precautionary assumption of 5% was used for this assessment.

3.4.1.3 HRA Project alone

136. No works for the proposed Norfolk Vanguard project will take place within the Alde-Ore Estuary SPA site boundary. The main potential impact for lesser black-backed gull is therefore in relation to collision risk when birds are outside of the SPA site boundary; these gulls fly partly within the height range where they may encounter rotating turbine blades.

137. The predicted monthly numbers of lesser black-backed gull collision mortalities based on Band Option 2 (Band 2012), with an avoidance rate of 99.5% (the avoidance rate as agreed with Natural England for use in Band model Option 1 or 2 collision risk modelling) for the proposed Norfolk Vanguard project, are shown in Table 18.

Table 18. Predicted monthly numbers collision estimates for lesser black-backed gull at the Norfolk Vanguard site calculated using Band Option 2 (generic flight heights) for the worst case turbine option (10MW).

Month	Deterministic collision mortality (mean density and 95% c.i.)	Monthly proportions (assumed 17% breeding season, 3.3% migration periods and 5% in mid-winter; see section 3.4.1.2)
January	0.82 (0-1.89)	0.04 (0-0.09)

Month	Deterministic collision mortality (mean density and 95% c.i.)	Monthly proportions (assumed 17% breeding season, 3.3% migration periods and 5% in mid-winter; see section 3.4.1.2)
February	0.22 (0-0.55)	0.01 (0-0.03)
March	0.56 (0-2.23)	0.02 (0-0.07)
April	**0.67 (0-2.15)**	**0.11 (0-0.37)**
May	**0 (0-0)**	**0 (0-0)**
June	**3.03 (0.43-6.5)**	**0.52 (0.07-1.11)**
July	**4.22 (1.72-8.02)**	**0.72 (0.29-1.36)**
August	**7.65 (2.82-13.78)**	**1.3 (0.48-2.34)**
September	2.5 (0.78-4.67)	0.08 (0.03-0.15)
October	2.75 (0.76-5.98)	0.09 (0.03-0.2)
November	0.33 (0-0.89)	0.02 (0-0.04)
December	0.29 (0-0.72)	0.01 (0-0.04)
Total	23.05 (6.51-47.38)	2.9 (0.9-5.8)

Months in bold indicate the full breeding months (note that the migration free breeding season has also considered in the assessment).

138. The majority of collisions are predicted during the second half of the breeding season and early autumn (June to August). This indicates wider movements of failed and nonbreeding individuals and birds on migration through the southern North Sea.

139. During the migration-free breeding season (May to July) the total number of predicted collisions was 7.2 (14.5using the upper 95% confidence interval), while for the full breeding season this figure was 18.1 (35.1 using the upper 95% confidence interval). On the basis of the seasonal percentages of Alde-Ore SPA birds predicted to be on the Norfolk Vanguard site (figures derived above), using the full breeding season would be up to 2.9 birds (Table 19).

Table 19. Estimated Alde-Ore lesser black-backed gull collision risk at Norfolk Vanguard calculated using deterministic collision estimates and seasonal percentages as detailed in the text.

Month	Migration free breeding season		Full breeding season	
	Total	Alde-Ore	Total	Alde-Ore
Spring (3.3%)	1.23	0.13	0.56	0.02
Breeding season (17%)	7.25	1.23	15.57	2.65
Autumn (3.3%)	12.90	1.47	5.25	0.17
Winter (5%)	1.66	0.08	1.66	0.08
Total	**23.04**	**2.92**	**23.04**	**2.92**

140. Natural mortality for the SPA population (assuming approximately 4,000 adults) would be around 460 individuals at an average adult mortality rate of 11.5%

(Horswill and Robinson 2015). A total additional worst case mortality of up to 2.9 birds (using the full breeding season) due to collisions at the Norfolk Vanguard site would increase the mortality rate by 0.6. Using the upper 95% confidence interval (5.8) this increase would be 1.3% and using the lower 95% confidence interval this would be 0.2%.

141. Following SNCB recommendations, an increase in mortality of less than 1% is considered to be undetectable against the range of background variation. While the upper 95% confidence interval estimate slightly exceeds the 1% threshold of detectability, the margin above the threshold equates to one individual (i.e. one less mortality per year brings the prediction below the 1% threshold). Therefore, since the increased mortality predicted as a result of mean collisions at Norfolk Vanguard is below the agreed threshold, at which increases in mortality are detectable, and the upper confidence interval only just exceeds this level it is reasonable to conclude that there will be no adverse effect on the integrity of the Alde-Ore Estuary SPA as a result of lesser black-backed gull collisions at the proposed Norfolk Vanguard project alone.

3.4.1.4 EIA cumulative and HRA In-combination

142. The cumulative lesser black-backed gull collision risk prediction has been calculated for all wind farms in the North Sea (Table 20).

Table 20. Lesser black-backed gull collision mortality for all wind farms (nonbreeding) and those with potential connectivity during the breeding season with the Alde-Ore SPA.

Wind farm	Predicted collisions (@ 99.5% avoidance rate, Band Model option 2)			
	Annual	Nonbreeding	Breeding (Annual minus nonbreeding)	Projects within 141km of Alde Ore SPA
Beatrice Demonstrator	0.0	0.0	0.0	0
Greater Gabbard	62.0	49.6	12.4	12.4
Gunfleet Sands	1.0	0.0	1.0	1.0
Kentish Flats	1.6	1.3	0.3	0.3
Lincs	8.5	6.8	1.7	0
London Array	0.0	0.0	0.0	0
Lynn and Inner Dowsing	0.0	0.0	0.0	0
Scroby Sands	0.0	0.0	0.0	0
Sheringham Shoal	8.3	6.6	1.7	1.7
Teesside	0.0	0.0	0.0	0
Thanet	16.0	12.8	3.2	3.2
Humber Gateway	1.3	1.1	0.3	0
Westermost Rough	0.3	0.3	0.1	0
Hywind	0	0	0	0
Kincardine	0	0	0	0

Wind farm	Predicted collisions (@ 99.5% avoidance rate, Band Model option 2)			
	Annual	Nonbreeding	Breeding (Annual minus nonbreeding)	Projects within 141km of Alde Ore SPA
Beatrice	0.0	0.0	0.0	0
Dudgeon	38.3	30.6	7.7	7.7
Galloper	138.8	111.0	27.8	27.8
Race Bank	54.0	10.8	43.2	0
Rampion	7.9	6.3	1.6	0
Hornsea Project One	21.8	17.4	4.4	0
Blyth Demonstration Project	0.0	0.0	0.0	0
Dogger Bank Creyke Beck Projects A and B	13.0	10.4	2.6	0
East Anglia ONE	39.7	33.8	5.9	5.9
European Offshore Wind Deployment Centre	0.0	0.0	0.0	0
Firth of Forth Alpha and Bravo	10.5	8.4	2.1	0
Inch Cape	0.0	0.0	0.0	0
Moray Firth (EDA)	0.0	0.0	0.0	0
Neart na Gaoithe	1.5	1.2	0.3	0
Dogger Bank Teesside Projects A and B	12.0	9.6	2.4	0
Triton Knoll	37.0	29.6	7.4	0
Hornsea Project Two	4.0	2.0	2.0	0
East Anglia THREE	10.0	8.2	1.8	1.8
Hornsea Project Three	17.3	0	17.3	0
Thanet Extension	2.3	0.8	1.5	1.5
Moray West	0	0	0	0
Norfolk Vanguard	23.0	7.5	15.6	2.9
Total (inc. Hornsea Project Three)	**530.1**	**366.1**	**164.3**	**66.2**
Total (exc. Hornsea Project Three)	**512.8**	**366.1**	**147**	**66.2**

3.4.1.4.1 Cumulative assessment

143. On the basis of the worst case Norfolk Vanguard collision estimates the annual cumulative total is 530.1 including Hornsea Project Three and 512.8 without this project.

144. The background mortality for the largest BDMPS population (209,007) at an all age class average mortality rate of 0.141 is 26,335. The addition of 530.1 to this increases the rate by 1.8%, and without Hornsea Project Three this would be 1.7%.

145. Note, however that many of the collision estimates for other wind farms were calculated on the basis of consented designs with higher total rotor swept areas than have been installed (or are planned), which is a key factor in collision risk. For example, the Galloper wind farm, which is currently under construction, was consented on the basis of 140 turbines but only 56 have been installed. A method

for updating collision estimates for changes in wind farm design was presented in Trinder (2017). Updating the collision estimates for the Galloper wind farm using this approach reduces the predicted annual mortality from 139 to 60. Applying the same method to the other wind farms in Table 20 can achieve a reduction in the cumulative annual mortality of around 200. Therefore, the values presented in Table 20, as well as being based on precautionary calculation methods, can be seen to overestimate the total risk by around 35% due to the reduced collision risks for projects which undergo design revisions post consent.

146. Lesser black-backed gull collision assessments undertaken prior to 2014 were made on the basis of Band model Option 1 and an avoidance rate of 98%, with the change to 99.5% dating from November 2014 (JNCC *et al.,* 2014). Therefore, projects consented prior to this date were on the basis of a cumulative collision mortality 4 times that presented in Table 20. Accounting for projects up to Triton Knoll consented after November 2014 (Hornsea Project 1, 22 annual collisions at 99.5%; Dogger Bank Creyke Beck A&B, 13 annual collisions at 98.9% Option 3; Dogger Bank Teesside A&B, 12 annual collisions at 98.9% Option 3) the previous cumulative collision total (at 98%) excluding these three projects would have been 1,656 (461 – (22+13+12) x 4). The current worst case cumulative total of 530.1, including all consented and still to be consented projects, is therefore much lower than this previously accepted cumulative total. Indeed, even if all of the previous consents had been granted on the basis of an avoidance rate of 99% this would still be around 828, 1.5 times the current cumulative prediction. The same approach can be applied to the seasonal estimates, which are all lower than the cumulative totals for the projects granted consent in 2014.

147. A review of nocturnal activity in seabirds (EATL, 2015) has indicated that the value currently used for this parameter (50%, as used for estimating the collisions at Norfolk Vanguard detailed above) to estimate collision risk at night for lesser black-backed gull is almost certainly an overestimate, possibly by as much as a factor of two (i.e. study data suggest that 25% is more appropriate). This study found that reducing the nocturnal activity factor to 25% reduces collision estimates by around 15%. This adjustment to nocturnal activity is also applicable to the other cumulative collision estimates in Table 20. A correction applied by this method would reduce the overall collision estimate for all wind farms by a significant amount (e.g. between 7% and 25%; note the magnitude of reduction varies depending on the time of year and wind farm latitude due to the variation in day and night length). This further emphasises the precautionary nature of the current assessment.

148. In conclusion, the current cumulative total is considerably lower than previously consented cumulative totals (between 1.5 and 3 times lower), and yet this total still includes several sources of precaution (e.g. consented vs. built impacts and

overestimated nocturnal activity). Therefore, the cumulative impact on the lesser black-backed gull population due to collisions both year round and within individual seasons is considered to be of low magnitude and lesser black-backed gull are considered to be of low sensitivity, therefore the impact significance is **minor adverse**.

3.4.1.4.2 In-combination assessment

149. The total breeding season lesser black-backed gull collision estimate is 164.3, including Hornsea Project Three and 147.0 without this project. Given that tracking studies have revealed low connectivity for the Alde-Ore SPA population with the Norfolk Vanguard site (Thaxter et al. 2012b, 2015), it is questionable both whether the proposed Norfolk Vanguard project would contribute to an in-combination total during the breeding season, and also if all of the wind farms within 141 km should be considered. However, as a precautionary assessment with respect to the Alde-Ore SPA population, wind farms within 141 km of the Alde-Ore SPA have been considered during the breeding season, on the grounds that only these wind farms have the potential to contribute to mortality on the SPA population at this time of year. Hence the breeding season mortality has been summed for Greater Gabbard, Gunfleet Sands, Kentish Flats, London Array, Scroby Sands, Sheringham Shoal, Thanet, Thanet Extension, Dudgeon, East Anglia ONE, Galloper and East Anglia THREE. The total breeding season mortality for these wind farms is 66.2, to which Norfolk Vanguard adds 2.9. However, it is more likely that the breeding season total should be based on wind farms within the mean foraging range of 72 km (Greater Gabbard, East Anglia ONE, Galloper, London Array) which indicate a total breeding season mortality estimate of 45 collisions.

150. Allowing for the relative size of the Alde Ore Estuary SPA population compared with that in Norfolk and Suffolk as a whole within 141 km of the SPA (the SPA is estimated to represent 30% of the total Norfolk and Suffolk lesser black-backed gull population, as discussed above), the breeding season total was estimated to be 19.9 (30% of the other wind farm total of 63.3 plus 2.9 at Norfolk Vanguard).

151. In the nonbreeding season, as discussed above, given the large geographical area from which lesser black-backed gulls migrating through the Norfolk Vanguard site originate, it is only possible to apportion mortality to the Alde-Ore SPA population on the basis of its size relative to the wider lesser black-backed gull population. Across all age classes the Alde-Ore Estuary SPA represents approximately 3.3% of the BDMPS autumn population, about 3.3% of the BDMPS spring population and a maximum of 5% of the BDMPS winter population. As noted above, for many wind farms there is insufficient information to determine in which months nonbreeding season collisions occur. Therefore, on the basis of the whole period a weighted Alde-

Ore Estuary SPA percentage of 4% has been calculated (5 months at 3.3% and 4 months at 5%). This indicates that up to 15 birds (366 x 4%) from the Alde-Ore Estuary SPA population could be at risk of collision during the nonbreeding season (of which 0.3 are attributed to Norfolk Vanguard).

152. The annual mortality of lesser black-backed gulls from the Alde-Ore SPA is therefore 15 during the nonbreeding season and 19.9 during the breeding season, 35 in total (of which Norfolk Vanguard contributes up to 2.9).

153. In-combination mortality of up to 35 birds attributable to the Alde-Ore SPA population of lesser black-backed gulls compares with estimated natural mortality of about 460 birds per year. Thus, the additional in-combination mortality would increase the mortality rate by 7.6%.

154. Recent work has highlighted the reduction in collisions which results from updating consented assessments to reflect as-built wind farm designs in comparison to the original full consent envelopes (Trinder 2017). For the wind farms within foraging range of Alde Ore Estuary SPA where this has been undertaken updating from the consented design to the as-built design reduces predicted mortality by an average of 33% (Trinder 2017), which would reduce the in-combination mortality prediction for existing wind farms from 19 (63.3 x 0.3, accounting for the SPA proportion of birds present) to around 12.7 (19 x 0.67, accounting for headroom reduction), to which the Norfolk Vanguard project adds 2.9 (15.6 in total). The same reduction applied to the nonbreeding estimate of 15 would reduce this to 10. Therefore, the annual mortality would be 25.6 which would result in an increase in background mortality of 5.5%.

155. To provide context for these estimates, it is worth noting that the in-combination collision total predicted for the Galloper Wind Farm was 85 when this wind farm was consented (using the methods recommended at that time but updated to the 99.5% avoidance rate to ensure comparability), which is more than double the more precautionary estimate of 35 above, and more than three times the more likely prediction of 25.6.

156. It is also worth noting the comments made by the Secretary of State in relation to the East Anglia ONE assessment. Despite the much lower avoidance rate applied at the time of that assessment (98%), it was concluded by the Secretary of State in relation to East Anglia ONE (DECC 2014), that the mortality from offshore wind farms is insignificant compared to other factors affecting the population of the lesser black-backed gull, and with planned improvements to the SPA (such as excluding predatory mammals from gull colonies), immigration from other colonies is likely, and would boost numbers, should favourable breeding conditions be created.

157. To summarise the above calculations, the adult annual, in-combination mortality predictions are:

- 35 (based on 141 km foraging range) comprising:
  - 15 nonbreeding (0.3 at Norfolk Vanguard),
  - 19.9 breeding (63.3 x 0.3 for other wind farms within 141km plus 2.9 at Norfolk Vanguard);
- 30 (based on 72 km foraging range) comprising:
  - 15 nonbreeding (0.3 at Norfolk Vanguard),
  - 15 breeding if wind farms within 72km are included in the breeding season (45 x 30% accounting for the Alde Ore Estuary SPA percentage of the Norfolk and Suffolk population with potential connectivity; 0 at Norfolk Vanguard);
- 25.6 (based on 141 km foraging range and consent vs. built reduction) comprising:
  - 10 nonbreeding (15 x 67%; 0.3 at Norfolk Vanguard),
  - 15.6 breeding (63.3 x 0.3 x 0.67 of the existing wind farm total plus 2.9 at Norfolk Vanguard).

158. A population model was developed to provide further interpretation of these potential in-combination impacts (MacArthur Green 2019). This model was developed following current NE guidance, utilising a matched-run approach to generate counterfactuals of population size (CPS) and counterfactuals of population growth rate (CPGR) and run for a simulated period of 30 years. Summary results are provided in Table 21.

Table 21. Lesser black-backed gull Alde Ore Estuary SPA population modelling results (see MacArthur Green 2019 for details).

Model	Adult mortality	Counterfactual metric (after 30 years)		Source table (Appendix 1)
		Growth rate	Population size	
Density independent	25	0.991	0.833	Tables A.1 & A.2
	40	0.987	0.687	
Density dependent	25	0.997	0.0.947	Tables A.3 & A.4
	40	0.996	0.914	

159. Taking the modelled adult mortality of 40 (as the worst case), the population growth rate was predicted to be 1.3% lower (0.987) than the baseline using the density independent model, and 0.4% lower (0.996) using the density dependent model. At

the lower modelled adult mortality of 25, the reduction in growth rate was 0.9% for the density independent model and 0.3% for the density dependent model.

160. Although there is a lack of reliable evidence on the population trend at the SPA since 2010 (the last all SPA count available), the predicted reductions in growth rate, which only just exceed 1% for the most precautionary combination of worst case collisions (including the use of the much higher mortality predictions estimated for consented wind farm designs rather than for the as built designs and over-estimated nocturnal activity) and worst case modelled predictions, are considered very unlikely to have a detectable effect on the population.

161. The more realistic collision estimates, accounting for the reduced impacts from built wind farms compared with the consented designs, predict a growth rate reduction of no more than 0.9% (density independent), which further reduces any concerns about the impact on the SPA population.

162. The relevant conservation objective is to restore breeding numbers of lesser black-backed gulls from the present level of about 2,000 pairs back to the population size at designation which was about 14,000 pairs. The annual number of predicted lesser black-backed gull collisions at the Norfolk Vanguard site, including the precautionary assumption of an extended breeding season, which can be attributed to the Alde Ore SPA is very small (no more than 2.9) and therefore not considered to materially alter the natural mortality rate for this population. Therefore, no adverse effect on the integrity of the Alde-Ore SPA lesser black-backed gull population is predicted as a result of the proposed Norfolk Vanguard project alone.

163. Therefore, the conclusions presented in the Norfolk Vanguard ES and HRA and subsequent submissions (ExA; AS; 10.D6.17) remain the same; it can be concluded that there will be no adverse effect on the integrity of Alde-Ore Estuary SPA from collision impacts on lesser black-backed gull due to the proposed Norfolk Vanguard project in-combination with other plans and projects.

164. Furthermore, the contribution from Norfolk Vanguard to this total has been substantially reduced following design revisions to mitigate collision risks, with the annual collision mortality estimate for lesser black-backed gull reduced by 47% when the removal of the 9MW turbine, revised layout and turbine draught height increase are considered together.

165. Furthermore, the context for the status of this population is relevant to the significance of potential collision mortality. The breeding success, and hence the population trend, of lesser black-backed gulls in the Alde-Ore SPA population appears to be mainly determined by the amount of predation, disturbance and

flooding occurring at this site (Department of Energy and Climate Change 2013a, Thaxter et al. 2015). Increased predation and disturbance by foxes has been considered the main factor causing reductions in breeding numbers. Management measures to reduce access by foxes has resulted in some recovery of numbers of gulls. The main driver of gull numbers in this SPA therefore appears to be suitable management at the colonies to protect gulls from predators (Department of Energy and Climate Change 2013a). It seems apparent that further efforts in this regard could improve this population's conservation status.

3.5 Great black-backed gull

3.5.1 Collision risk

3.5.1.1 EIA Project alone

166. The revised collision risk estimates for great black-backed gull for the 10MW turbine and the revised project layout (ExA; CRM 10.D6.5.1) and 5m turbine draught height increase (ExA;AS;10.D.7.5.2), calculated using the Band (2012) deterministic model and Natural England's preferred parameter values, are provided in Table 22.

Table 22. Great black-backed gull seasonal and annual collision risk using the migration free (May to July) and full (March to August) breeding seasons.

Breeding season	Migration free breeding2.3	Midwinter/non-breeding	Annual
Migration-free	1.31 (0-3.49)	45.54 (22.68-74.6)	46.84 (22.68-78.08)
Full	8.09 (1.72-16.69)	38.76 (20.96-61.39)	

Note: No months are included in more than one season (overlapping months were assigned to the breeding season). Seasons from Furness (2015).

167. In the submission at Deadline 6.5 (ExA; CRM 10.D6.5.1, Table 2) it was concluded that a higher annual mortality of 61.9 (prior to the turbine draught height increase) would not increase the background rate by more than 1% and therefore it was concluded that Norfolk Vanguard alone would have no significant impact at the EIA scale. This conclusion is further supported by the lower revised annual estimate of 46.8 (a reduction of 24% for the draught height alone) and therefore the conclusion of no significant impact at the EIA scale remains valid.

3.5.1.2 EIA Cumulative

168. The cumulative great black-backed gull collision risk prediction is presented in Table 23. This collates collision predictions from other wind farms which may contribute to the cumulative total. This table takes the wind farm assessment for East Anglia THREE as its starting point and adds estimates for wind farms submitted since that project's application.

169. The collision values presented in Table 23 include totals for breeding, nonbreeding and annual periods. However, not all projects provide a seasonal breakdown of collision impacts, therefore it is not possible to extract data from these periods for cumulative assessment. Natural England has previously noted that an 80:20 split between the nonbreeding and breeding seasons is appropriate for lesser black-backed gull in terms of collision estimates (Natural England, 2013), and this has been used for great black-backed gull. Therefore, for those sites where a seasonal split was not presented the annual numbers in Table 23 have been multiplied by 0.8 to estimate the nonbreeding component and 0.2 to estimate the breeding component.

Table 23. Great black-backed gull cumulative collision risk.

Wind farm	Breeding	Nonbreeding	Annual
Beatrice Demonstrator	0.0	0.0	0.0
Greater Gabbard	15.0	60.0	75.0
Gunfleet Sands	0.0	0.0	0.0
Kentish Flats	0.1	0.2	0.3
Lincs	0.0	0.0	0.0
London Array	0.0	0.0	0.0
Lynn and Inner Dowsing	0.0	0.0	0.0
Scroby Sands	0.0	0.0	0.0
Sheringham Shoal	0.0	0.0	0.0
Teesside	8.7	34.8	43.6
Thanet	0.1	0.4	0.5
Humber Gateway	1.3	5.1	6.3
Westermost Rough	0.0	0.0	0.1
Hywind	0.3	4.5	4.8
Kincardine	0.0	0.0	0.0
Beatrice	30.2	120.8	151.0
Dudgeon	0.0	0.0	0.0
Galloper	4.5	18.0	22.5
Race Bank	0.0	0.0	0.0
Rampion	5.2	20.8	26.0
Hornsea Project One	17.2	68.6	85.8
Blyth Demonstration Project	1.3	5.1	6.3
Dogger Bank Creyke Beck Projects A and B	5.8	23.3	29.1
East Anglia ONE	0.0	46.0	46.0
European Offshore Wind Deployment Centre	0.6	2.4	3.0
Firth of Forth Alpha and Bravo	13.4	53.4	66.8
Inch Cape	0.0	36.8	36.8
Moray Firth (EDA)	9.5	25.5	35.0
Neart na Gaoithe	0.9	3.6	4.5
Dogger Bank Teesside Projects A and B	6.4	25.5	31.9
Triton Knoll	24.4	97.6	122.0
Hornsea Project Two	3.0	20.0	23.0
East Anglia THREE	4.6	34.4	39.0

Wind farm	Breeding	Nonbreeding	Annual
Hornsea Project Three*	19.4	46.6	66.0
Thanet Extension	1.3	20.8	22.1
Moray West	4.0	5.0	9.0
Norfolk Vanguard	8.1	38.8	46.8
Total (inc. Hornsea Project Three)	**185.3**	**818**	**1003.2**
Total (exc. Hornsea Project Three)	**165.9**	**771.4**	**937.2**

170. On the basis of the worst case Norfolk Vanguard collision estimate the annual cumulative total including Hornsea Project Three is 1,003.2 and without this project is 937.2.

171. The background mortality for the largest BDMPS population (91,399) at an all age class average mortality rate of 0.185 is 16,909. The addition of 1018.2 to this increases the rate by 5.9%, and without Hornsea Project Three this would be 5.5%.

172. Many of the collision estimates for other wind farms were calculated on the basis of consented designs with higher total rotor swept areas than have been installed (or are planned), which is a key factor in collision risk. For example, the Beatrice wind farm, which is currently under construction, was consented on the basis of 125 turbines but only 84 are being installed. A method for updating collision estimates for changes in wind farm design was presented in Trinder (2017). Updating the collision estimates for the Beatrice wind farm using this approach reduces the predicted annual mortality from 151 to 101. Applying the same method to the other wind farms in Table 23 can achieve a reduction in the cumulative annual mortality of around 260. Therefore, the values presented in Table 23, as well as being based on precautionary calculations, can be seen to overestimate the total risk by around 30% due to the reduced collision risks for projects which undergo design revisions post consent.

173. Great black-backed gull collision assessments undertaken prior to 2014 were made on the basis of Band model Option 1 and an avoidance rate of 98%, with the change to 99.5% dating from November 2014 (JNCC *et al.,* 2014). Therefore, projects consented prior to this date were on the basis of a cumulative collision mortality 4 times that presented in Table 23. Accounting for projects up to Triton Knoll consented after November 2014 (Hornsea Project One, 86 annual collisions at 99.5%; Dogger Bank Creyke Beck A&B, 29 annual collisions at 98.9% Option 3; Dogger Bank Teesside A&B, 32 annual collisions at 98.9% Option 3) the previous cumulative collision total (at 98%) excluding these three projects would have been 2,524 (778 - (86 + 29 + 32) x 4. The current worst case cumulative total of 1,018.2, including all consented and still to be consented projects, is therefore much lower than the previously accepted cumulative total. Indeed, even if all of the previous consents

had been granted on the basis of an avoidance rate of 99% this would still be higher than the current cumulative prediction. The same approach can be applied to the seasonal estimates, which are all lower than the cumulative totals for the projects granted consent in 2014.

174. A review of nocturnal activity in seabirds (EATL, 2015) has indicated that the value currently used for this parameter (50%) to estimate collision risk at night for great black-backed gull is almost certainly an overestimate, possibly by as much as a factor of two (i.e. study data suggest that 25% is more appropriate). Reducing the nocturnal activity factor to 25% reduced collision estimates by around 15%. This adjustment to nocturnal activity is also applicable to the other cumulative collision estimates. A correction applied by this method would reduce the overall collision estimate for all wind farms by a significant amount (e.g. between 7% and 25%; note the magnitude of reduction varies depending on the time of year and wind farm latitude due to the variation in day and night length). This further emphasises the precautionary nature of the current assessment.

175. In the decision for the Rampion wind farm (Planning Inspectorate, 2014a; DECC, 2014), the cumulative collision mortality for great black-backed gull was considered. In their recommendations to the Secretary of State (Planning Inspectorate, 2014), the Examining Authority reported the cumulative mortality for this species as either 1,803 individuals per year (Applicant's estimate) or 3,025 (Natural England's estimate). The difference in these two values remained unresolved between the applicant and Natural England, however the Examining Authority (Planning Inspectorate, 2014) concluded:

'that the addition of Rampion OWF does not tip the balance in terms of exceeding a threshold that would not otherwise be exceeded.'

(Note that the threshold referred to in the above quote was the PBR value for this species, although PBR is no longer considered an appropriate tool for assessing wind farm impacts).

176. The current cumulative mortality of 1,003.2 is much lower than either of the cumulative totals reported for Rampion (1,803 and 3,025). The increase in the avoidance rate for this species has resulted in a large reduction in predicted cumulative totals to the extent that the current estimate is much lower than those on which it has been concluded there will be no effect on the population in the long term (DECC, 2014).

177. A population model for great black-backed gull was developed to inform the East Anglia THREE assessment (EATL 2016a). Four versions of the model were presented, using two different sets of demographic rates (from the literature) and both with

and without density dependent regulation of reproduction. Comparison of the historical population trend with the outputs from these models indicated that the density dependent versions generated population predictions which were much more closely comparable to the population trend. The density dependent models were also less sensitive to which set of demographic rates was used. The density dependent versions were therefore considered to provide a more reliable predictive tool.

178. Using the density dependent model, application of an additional annual mortality of 1,000 to the great black-backed gull BDMPS resulted in reductions in the population growth rate of up to 1.6% for the most precautionary density independent predictions (it should be noted that this was estimated across a period of 25 years, however the difference in growth rate changes across this period and that for a 30 year period will be small and would not alter the conclusion that this level of mortality would not have a significant effect on the long term growth rate of the population).

179. On the basis of the results from the modelling Natural England concluded that whilst a significant cumulative effect could not be ruled out, the project's (East Anglia THREE) individual contribution was so small that it would not materially affect the overall cumulative impact magnitude. It is also worth reiterating that the current cumulative total is considerably lower (due to the lower avoidance rates applied to this species in the past) than that which would have been estimated for older wind farm projects for which consent was granted.

180. The final East Anglia THREE annual collision impact for great black-backed gull was 39, which is only slightly lower than that for Norfolk Vanguard (46.8) . And as noted above, there are several sources of precaution involved in reaching this estimate (e.g. over-estimates of nocturnal activity and use of predictions for consented rather than built wind farm designs) and therefore it is reasonable to assume that the same conclusion (that Norfolk Vanguard's contribution will not materially alter the overall cumulative impact magnitude) would apply for the current project.

181. In conclusion, the cumulative impact on the great black-backed gull population due to collisions both year round and within individual seasons is considered to be of low magnitude and great black-backed gull are considered to be of low to medium sensitivity, therefore the impact significance is **minor adverse**.

3.6 Little gull

3.6.1 Collision risk

3.6.1.1 EIA Project alone

182. The revised collision risk estimates for little gull for the 10MW turbine and the revised project layout (ExA; CRM 10.D6.5.1)) and 5m turbine draught height increase (ExA;AS;10.D.7.5.2), calculated using the Band (2012) deterministic model and Natural England's preferred parameter values, are provided in Table 24.

Table 24. Little gull seasonal and annual collision risk.

Breeding season	Nonbreeding season	Annual
1.95 (0.65-3.57)	3.14 (1.06-6.09)	5.09 (1.71-9.66)

183. Considering the reduction in mortality due to the removal of the 9MW turbine, the revised layout and the increase in turbine draught height together, the predicted collision mortality for little gull at Norfolk Vanguard has been reduced by 71%.

184. In the Norfolk Vanguard HRA (Vattenfall 2018) the little gull population with connectivity to the southern North Sea was estimated to be up to 75,000 (Steinen et al. 2007), with a precautionary estimate of between 10,000 and 20,000 based on the surveys conducted across the Greater Wash Area of Search (a larger area than the SPA within which surveys were conducted to inform the spatial extent of the SPA) and analysis of those data in Natural England and JNCC (2016).

185. The adult survival rate for little gull is reported as 0.8 (Horswill and Robinson 2015). Therefore, the natural mortality of the population will vary between 2,000 and 15,000 (for populations of 10,000 and 75,000, respectively). An addition of 5.1 mortalities to these would increase the mortality rate by 0.25% and 0.03% respectively. These are less than the 1% threshold below which impacts are considered undetectable against background changes and therefore the magnitude of collision impacts at the EIA scale for Norfolk Vanguard alone is negligible and the impact is **minor adverse**.

3.6.1.2 HRA Project alone

186. Since the Norfolk Vanguard Offshore Wind Farm is wholly outside the Great Wash SPA boundary, for assessment of potential impacts, it is appropriate to consider the wider population in the southern North Sea of which the SPA population is a component. This was presented in the HRA submitted for Norfolk Vanguard (Vattenfall 2018, section 6.1.3.2) and the population estimates thus derived were summarised above (precautionary estimates of 10,000 to 20,000; note also that in

their comments on the Norfolk Vanguard HRA, Natural England agreed with the approach to estimating population sizes and apportioning, Natural England 2018).

187. The Greater Wash SPA designated population of little gull is 1,255, which is 12.6% of the most precautionary population estimate of 10,000 or 6.3% of a population of 20,000. On this basis, and assuming collisions would be distributed uniformly throughout the population, this would imply that a maximum of 0.6 individuals from the Greater Wash SPA population of little gull would be at risk of mortality due to collisions (12.6% of 5.1), which would be reduced further to 0.3 on the basis of the more realistic wider population (of 20,000). For the SPA population of 1,255, and assuming the wider population is 10,000, the addition of 0.6 individuals would increase the background mortality rate by 0.24%, while using the more realistic wider population estimate of 20,000 this increase in mortality rate would be 0.12%.

188. Thus, it can be concluded that the maximum additional mortality of one individual from the SPA population will be undetectable and there will be no adverse effect on the integrity of the Greater Wash SPA as a result of collisions at the Norfolk Vanguard project alone.

3.6.1.3 HRA In-combination

189. The predicted mortality of little gull at Norfolk Vanguard in-combination with other wind farms with potential connectivity to the Greater Wash SPA little gull population is 60.1 (Table 25).

Table 25. Assessed collision rates and updated little gull collision predictions for offshore wind farm sites with potential connectivity to the Greater Wash SPA.

Wind farm	Annual collisions	Avoidance rate (%)	Assessed wind farm size	Collisions updated for 99.2% avoidance rate	Built or proposed wind farm size	Collisions updated for built or proposed wind farm
Triton Knoll	65	98	288 * 3.6MW	26	TBC. c. 120	c. 15
Race Bank	52	98	206 * 3MW	21	91 * 6MW	12
Sheringham Shoal	8	98	108 * 3MW	3	88 * 3.6MW	3
Hornsea Project One	10	98	332 * 3.6MW	4	174 * 7MW	2
Hornsea Project Two	1.3	98	360 * 5MW	0.5	N/A	0.5
Hornsea Project Three	0.5	99.2	300 * 6MW	0.5	N/A	0.5
Norfolk Vanguard	5.1	99.2	180 * 10MW	5.1	N/A	5.1
In-combination total				**60.1**		**38.1**

190. Given a regional little gull population of between 10,000 and 20,000 this figure (60.1) represents an increase in background mortality of between 1.5% and 3.0% (although as noted above the population may be as large as 75,000, further reducing the magnitude of potential impact, to an increase in mortality of less than 0.4%). The Greater Wash SPA designated population of little gull is 1,255, which is 12.6% of a population of 10,000 or 6.3% of a population of 20,000. On this basis, and assuming collisions would be distributed uniformly throughout the population, this would imply that a maximum of 7.6 individuals from the Greater Wash SPA population would be at risk of in-combination collisions (12.6% of 60.1), although using the actual built projects (or planned designs) and noting that Triton Knoll has reduced its capacity to 90 turbines this would reduce to 4.8 individuals. Furthermore, the in-combination collisions would be reduced to 2.4 individuals on the basis of the more realistic wider population (of 20,000). These would give rise to increases in mortality for the SPA population of between 0.95% (2.4 individuals, for built projects and the realistic population of 20,000) and 3.0% using the most precautionary combination of consented development predictions and the smallest regional population estimate of 10,000 (7.6 individuals).

191. A very similar total collision estimate of 7 individuals was assessed by the Secretary of State (SoS) for the in-combination assessment for the Triton Knoll non-material change application (BEIS 2018). In relation to this estimate the SoS stated:

"Assuming collisions are attributed evenly amongst the regional population, this equates to 7 individuals from the Greater Wash population. Such a small impact would also be undetectable in the SPA population."

And also:

"in view of the small impacts quantified above, the Secretary of State considers that an Appropriate Assessment is not required in this case."

192. Thus, on the basis of an SPA in-combination mortality of 7.6, for the most precautionary interpretation of the potential risk to the population or a more realistic total of 2.4, the likelihood of an adverse effect on the integrity of the Greater Wash SPA population of little gull can be ruled out for the proposed Norfolk Vanguard project in-combination with other plans and projects.

3.7 Conclusion

193. This note provides updated cumulative and in-combination assessment for the Norfolk Vanguard Offshore Wind Farm following reductions in the project's predicted collision mortality risks achieved through design mitigations which included removal of the 9MW turbine from the design envelope, a limit to the

proportion of turbines which will be installed across the East and West sites and an increase of 5m in turbine draught height (from 22m to 27m). Together these mitigations have reduced the project collision risks by an average of 65% across all species, which results in a considerable reduction in the project's contribution to the cumulative and in-combination totals.

194. The conclusions of the collision risk assessments presented in the ES, HRA and updates submitted during the project's examination (ExA; AS; 10.D6.17, ExA; As;10.D7.21) remain unchanged, with no significant impacts predicted for collisions at the project alone or cumulatively and no predicted adverse effects on SPA integrity due to the project alone or in-combination with other plans or projects.

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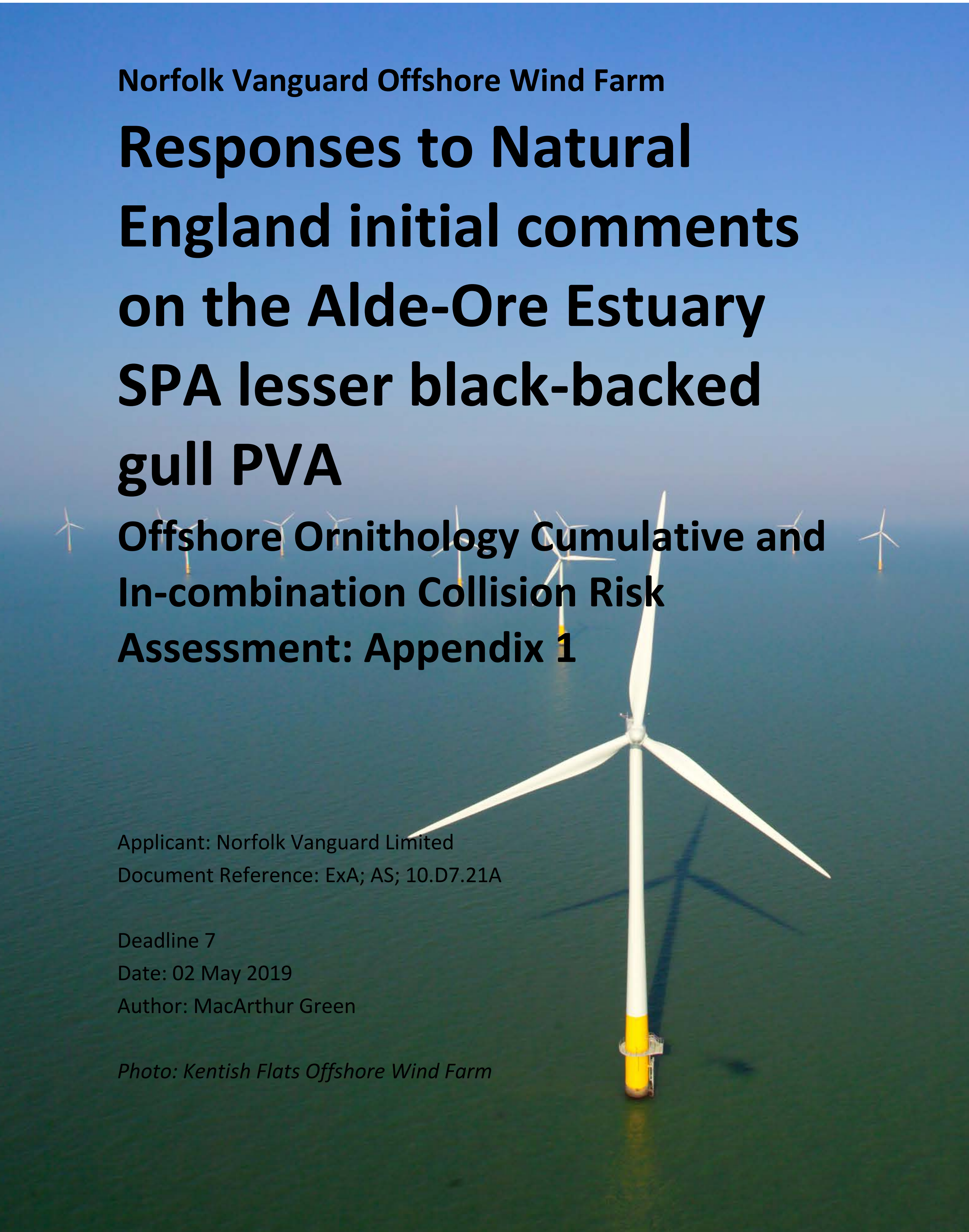
Norfolk Vanguard REP7-063: Alde Ore Estuary SPA PVA Responses

Norfolk Vanguard Limited Reference: ExA; AS; 10.D7.21: Cited in this document as MacArthur Green 2019b.



Norfolk Vanguard Offshore Wind Farm

Responses to Natural England initial comments on the Alde-Ore Estuary SPA lesser black-backed gull PVA

Offshore Ornithology Cumulative and In-combination Collision Risk Assessment: Appendix 1

Applicant: Norfolk Vanguard Limited
Document Reference: ExA; AS; 10.D7.21A

Deadline 7
Date: 02 May 2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
25/04/2019	01D	First draft for internal review	MT	EV/RS	MT
30/04/2019	02D	Minor revisions	MT	EV	
02/05/2019	04	Final for submission	MT	RS	RS

Executive Summary

This note contains responses from the Applicant to interim comments from Natural England on the Alde-Ore Estuary SPA lesser black-backed gull population viability analysis (PVA). This note also provides updated graphs of the counterfactuals of population size and population growth rate, estimated across 5,000 simulations and with the inclusion of 95% confidence intervals as requested by Natural England.

The outputs remain almost exactly the same for the purposes of assessment (i.e. within +/- 0.1% for the median predictions compared with those presented in ExA; AS; 10.D.6.16) and therefore the original interpretation in the Norfolk Vanguard assessment is unaffected by these updates.

Table of Contents

Glossary

CPGR	Counterfactual of Population Growth Rate
CPS	Counterfactual of Population Size
FFC	Flamborough and Filey Coast SPA
GWFL	Galloper Wind Farm Limited
LBBG	Lesser black-backed gull
OWF	Offshore Wind Farm
NE	Natural England
NV	Norfolk Vanguard
PVA	Population Viability Analysis
SMP	Seabird Monitoring Programme
SPA	Special Protection Area

1 INTRODUCTION

1. At Deadline 6, the Applicant submitted a population viability analysis (PVA) for the breeding population of lesser black-backed gulls at the Alde-Ore Estuary Special Protection Area (SPA) (ExA; AS; 10.D6.17) to provide predictions of the consequences of additional mortality from the Project on this population, as requested by Natural England.

2. Natural England provided the Applicant with initial comments on this PVA report via email on the 17th April 2019. Table 1 presents Natural England's comments and the Applicant's responses to these comments.

Table 1. Natural England's initial comments on the Alde-Ore Estuary SPA lesser black-backed gull PVA report (ExA; AS 10.D6.16) and responses from the Applicant.

Natural England's Comment	Applicant's Response
The models have been run using 1,000 simulations. We note that previous PVAs (e.g. MacArthur Green 2015) have used 5,000 simulations for the stochastic models, whereas the LBBG Alde-Ore PVA in REP6-020 undertaken by the Vanguard Applicant has used 1,000. As was advised by Natural England at Hornsea 3 regarding the updated PVAs undertaken for the Flamborough and Filey Coast (FFC) SPA, a larger number of simulations would potentially be needed to generate reliable results (Natural England 2019).	The Applicant acknowledges the point made by Natural England on this matter. While increasing the number of simulations as suggested, particularly when matched run formulations are used, makes virtually no material difference to the reliability of the results, updated outputs from the model with 5,00 simulations are provided in this note (figures 1 to 4 and tables 2 to 5). The median outputs for 5,000 simulations are within +/-0.1% of those produced for 1,000 simulations while the confidence intervals within +/-1% of those produced for 1,000 simulations, which makes no material difference to the conclusions reached from the results.
With regard to the metrics, it is not clear how the median and confidence intervals around the counterfactuals of population size and growth rate metrics have been calculated for the 'matched runs/pairs' approach. Therefore, Natural England suggests that the Applicant sets out how they have calculated the metrics - a worked example would be useful. Natural England advises that with a 'matched runs/pairs' method the metric should be calculated for each of the individual matched pairs and then (as there are 1,000 simulations in the Applicant's	The Applicant can confirm that the method described by Natural England is how these estimates were calculated and this is reflected in the values in the tabulated outputs of the report (ExA; AS; 10.D6.17, Tables A.1 to A.4). However, the confidence intervals were not

models) there will be 1,000 metric calculations from which a median value of the metric and the 95% confidence intervals can be derived.	presented on the counterfactual of population size figures (ExA; AS; 10.D6.17, Figures A.1 and A.3) and those intervals presented on the figures for the counterfactuals of population growth rate (ExA; AS; 10.D6.17, Figures A.2 and A.4) were incorrectly plotted. Updated figures are provided in this note (Figures 1 to 4 below) which present confidence intervals for both counterfactual measures which correspond to those in the tables in the report (A.1 to A.4) and which were estimated as per Natural England's methods. It should be noted that the median estimates in Figures 1 to 4 below are the same as those in the original report (ExA; AS; 10.D6.17).
We note that the final paragraph of Section 4 of REP6-020 states that: '…the demographic rates indicate that under baseline conditions the population growth rate would be in excess of 10%.' Natural England is concerned by this statement as there is no evidence to suggest this is an appropriate assumption. We note that in the original LBBG Alde-Ore PVA undertaken for the Galloper offshore wind farm (OWF) (GWFL 2012) when run in density independent mode and with the "historic" scenario resulted in projected population DECLINE - this was with: juvenile survival rate = 0.82, adult survival rate = 0.90, productivity = 0.45 chicks per pair and proportion of adults breeding = 0.66. These demographic rates are quite similar to the parameters used in this PVA undertaken for Vanguard (juvenile survival = 0.82, adult survival = 0.885, productivity = 0.53 and proportion of adults breeding = 0.663. Natural England does not think the Alde-Ore Estuary SPA colony is growing at all at the moment and therefore considers that its demographic rates must be different to those used here. Further justification for this assumption is needed should it continue to form part of the PVA.	This statement regarding baseline growth was made in error and reflected the results from an earlier draft of the model prior to demographic rate revisions to incorporate the relatively high level of nonbreeding recorded in this species. Following this update the underlying growth rate of the density independent model is negative (-2%). However, this has no bearing on the outputs presented and the counterfactual estimates are unaffected.
We note that the value of 0.351 fledged young per pair is a pretty low value. This figure has been arrived at by multiplying the Horswill & Robinson (2015) value of 0.530 for national mean productivity by 0.663 to take account of the proportion of birds that miss breeding each year (in an average LBBG population). Natural England is not certain about the appropriateness of this and note that in the old LBBG Alde-Ore PVA undertaken for Galloper OWF (GWFL 2012) three productivity rates were	The Applicant agrees this is a precautionary assumption regarding the incidence of nonbreeding, derived from other studies. However, the breeding success data suggested as alternatives by Natural England, only reflect birds which

simulated: 0.45, 0.80 and 1.0 and focused on the result when 0.8 was used. That was on the basis of there having been a good year in 2011. However, the 3 year mean productivity at Orford up to 2011 was 0.256 and in 2012 it was 0.19. We note that there is breeding success data in the Seabird Monitoring Programme (SMP) database for Havergate Island from 2009-11 and 2014-15, but no data for Orfordness.	actually attend the colony and initiate breeding, whereas the 66% figure accounts for birds which simply do not attempt to breed (i.e. do not attend the colony). This figure will not be known for this colony, although the Applicant agrees that given its small size it is possible that assuming such a high rate of nonbreeding rate as this (66%) overestimates the incidence of non-breeding. Thus, this ensures the results are precautionary, as reducing the rate of nonbreeding would improve the population's growth rate.
The last sentence of this paragraph states: 'Population projections produced by such models will either increase to infinity or decrease to extinction.' We note that if survival and productivity are perfectly matched then in theory the population may remain stable, but as the Applicant notes even if slightly out then over time the colony will drift up or down - though if quite closely matched the two stochastic elements may stop the inexorable rise or fall, or slow it considerably.	Natural England's comment on this matter would only be the case in a deterministic model, and even then the precision of the estimates to achieve stability in a density independent model is far beyond anything that could be estimated empirically. In a stochastic model such as the ones presented, the variation in parameter values means the original statement in the report remains correct.
We are not aware of any evidence of density dependence acting on the LBBG colony at the Alde-Ore Estuary SPA. The colony declined significantly in 2001, and although the reasons for the decline are not understood it may be due to external factors. It is now such a small colony that it is hard to imagine density dependence operating much now (unless maybe depensatory). This paragraph states: '…the demographic rate most likely to reflect density dependent effects will be reproduction, with breeding success declining as population approaches the ceiling set by food resources…' We note that density dependence will almost certainly NOT be operating just now at the Alde-Ore LBBG colony with such a depleted colony and will likely remain pretty weak effect until the colony gets much bigger. However, we consider it appropriate that the Applicant has considered modelling density dependent regulation through reproduction rather than survival across multiple rates.	The Applicant acknowledges Natural England's comment on this matter and agrees that modelling density dependence for seabirds is most appropriate through effects on reproduction. The Applicant considers that presenting outputs both with and without density dependence is appropriate in order that the range of potential population projections is available for assessment.
The last sentence of this paragraph states: 'Furthermore, the additional mortality was applied to all age classes in proportion to their presence (i.e. wind farm mortality was not considered	The Applicant can confirm that the mortality was applied using the model age ratios, not the

to target specific age classes).' Clarification is required as to whether this is in the modelled population as a whole or their presence in the OWF survey dataset of age classes recorded at sea. Natural England assumes it is the former, but clarification is required.	survey ones. The Applicant considers this to be appropriate because the modelling is intended to provide a guide for additional mortality in the wider sense (i.e. irrespective of where and when during the year it happens).
The first sentence of this paragraph states: 'Although the trend in the Alde-Ore Estuary population is not well known...' Natural England notes that the Alde-Ore LBBG population trend is well known from 2001 to 2010 at least, as shown in one of the figures in the Alde-Ore LBBG stochastic PVA report undertaken for Galloper OWF (GWFL 2012).	The Applicant stands by this statement: the trend since 2010 (i.e. the last decade) is not known with any confidence and the trend up to 2010 is not considered to provide a reliable guide for the current status of the population, as this is almost 10 years out of date (it is noted that this is a much longer gap than the two year gap when the analysis was undertaken for Galloper).

1.1 Updated outputs

3. Revised counterfactual figures are provided below calculated from 5,000 simulations as requested by Natural England, with 95% confidence intervals included on the figures. These results are within +/-0.1% for the median estimates and +/11% for the confidence intervals (compared with those obtained from 1,000 simulations (i.e. in ExA; AS; 10.D6.16). Therefore, the original interpretation of these results (i.e. in ExA; AS; 10.D6.17) is un affected. The corresponding outputs are tabulated in Tables 2 to 5.

4. The Applicant considers that this note addresses the comments received from Natural England and no further updates to the PVA outputs are therefore required.

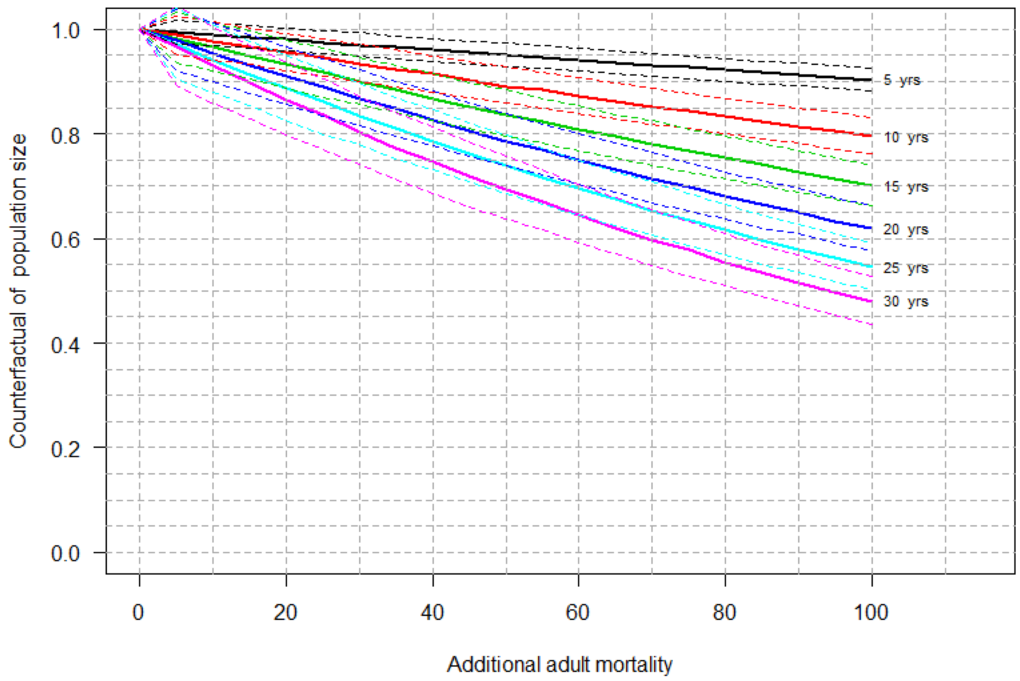


Figure 1. Counterfactual of population size, with 95% confidence intervals (dashed lines). 5,000 Density independent simulations.

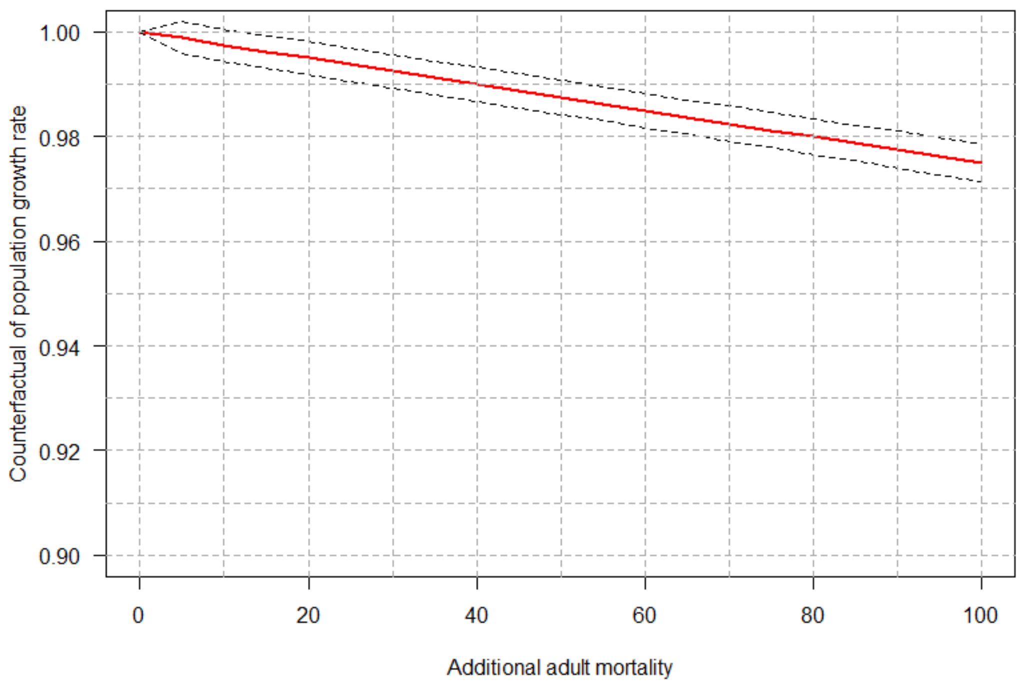


Figure 2. Counterfactual of population growth rate, with 95% confidence intervals (dashed lines). 5,000 Density independent simulations.

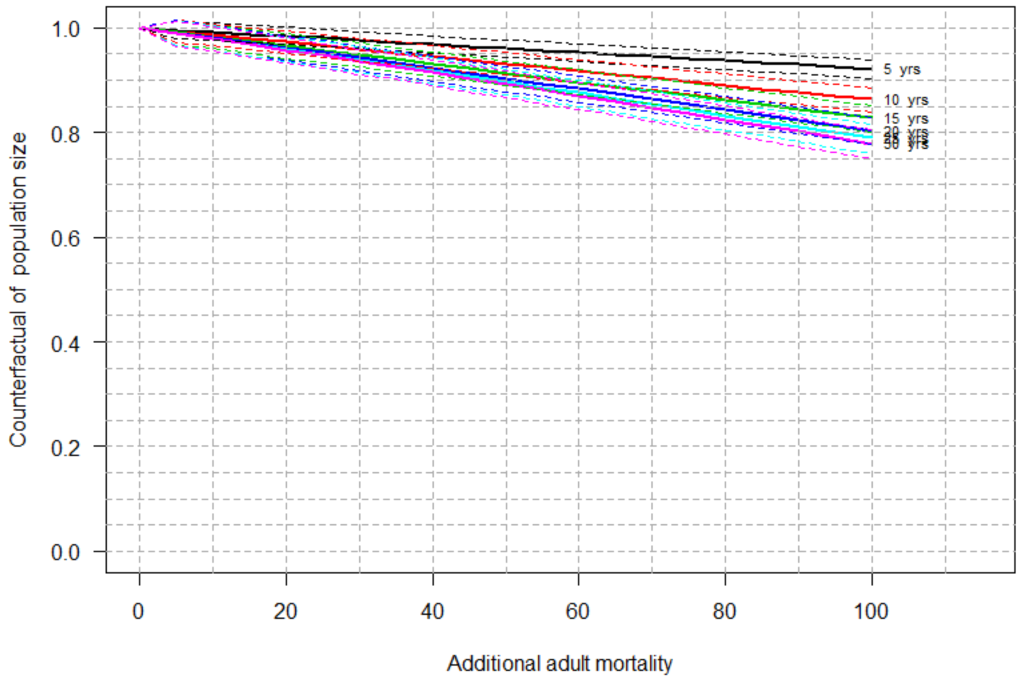


Figure 3. Counterfactual of population size, with 95% confidence intervals (dashed lines). 5,000 Density independent simulations.

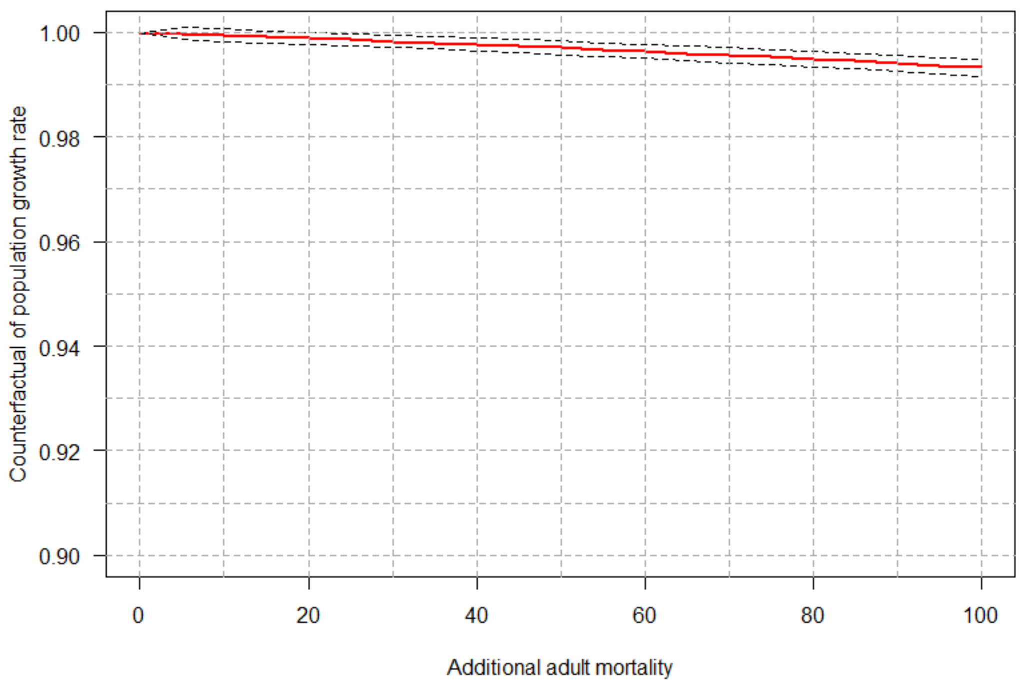


Figure 4. Counterfactual of population growth rate, with 95% confidence intervals (dashed lines). 5,000 Density dependent simulations.

Table 2. Lesser black-backed gull, demographic rate set 1, counterfactuals of population size after 5 to 30 years, estimated using a matched runs method, from 5,000 density independent simulations.

	Counterfactual of population size at 5 year intervals						
Additional adult mortality	**Estimate**	**yr.5**	**yr.10**	**yr.15**	**yr.20**	**yr.25**	**yr.30**
0	Lower 95%	1.000	1.000	1.000	1.000	1.000	1.000
	Median	1.000	1.000	1.000	1.000	1.000	1.000
	Upper 95%	1.000	1.000	1.000	1.000	1.000	1.000
5	Lower 95%	0.972	0.952	0.937	0.922	0.906	0.893
	Median	0.995	0.988	0.983	0.977	0.972	0.966
	Upper 95%	1.017	1.026	1.032	1.038	1.041	1.046
10	Lower 95%	0.968	0.942	0.920	0.899	0.879	0.858
	Median	0.990	0.978	0.966	0.953	0.941	0.930
	Upper 95%	1.014	1.016	1.012	1.012	1.009	1.006
15	Lower 95%	0.963	0.932	0.905	0.878	0.854	0.828
	Median	0.985	0.967	0.949	0.931	0.914	0.897
	Upper 95%	1.007	1.003	0.997	0.989	0.978	0.969
20	Lower 95%	0.959	0.922	0.888	0.857	0.827	0.798
	Median	0.980	0.956	0.932	0.910	0.887	0.865
	Upper 95%	1.002	0.991	0.979	0.966	0.953	0.940
25	Lower 95%	0.953	0.909	0.871	0.835	0.799	0.769
	Median	0.975	0.945	0.916	0.888	0.860	0.833
	Upper 95%	0.998	0.982	0.964	0.944	0.923	0.906
30	Lower 95%	0.948	0.900	0.856	0.815	0.780	0.741
	Median	0.971	0.934	0.900	0.866	0.834	0.804
	Upper 95%	0.994	0.972	0.947	0.923	0.898	0.872
35	Lower 95%	0.943	0.889	0.841	0.795	0.753	0.714
	Median	0.965	0.924	0.884	0.846	0.810	0.775
	Upper 95%	0.988	0.961	0.932	0.901	0.870	0.843
40	Lower 95%	0.940	0.879	0.827	0.778	0.731	0.687
	Median	0.961	0.914	0.869	0.826	0.785	0.748
	Upper 95%	0.982	0.949	0.914	0.881	0.846	0.815
45	Lower 95%	0.935	0.870	0.811	0.758	0.708	0.662
	Median	0.956	0.903	0.854	0.807	0.762	0.719
	Upper 95%	0.978	0.939	0.898	0.859	0.821	0.785
50	Lower 95%	0.929	0.859	0.796	0.740	0.687	0.638
	Median	0.951	0.893	0.838	0.787	0.739	0.694
	Upper 95%	0.974	0.929	0.884	0.840	0.799	0.757
55	Lower 95%	0.925	0.849	0.783	0.721	0.666	0.616
	Median	0.946	0.883	0.824	0.769	0.717	0.669
	Upper 95%	0.968	0.917	0.867	0.820	0.774	0.731
60	Lower 95%	0.920	0.840	0.767	0.704	0.645	0.592
	Median	0.941	0.873	0.809	0.751	0.696	0.645
	Upper 95%	0.963	0.907	0.853	0.800	0.749	0.703
65	Lower 95%	0.915	0.829	0.756	0.687	0.626	0.571

	Median	0.937	0.863	0.795	0.732	0.675	0.622
	Upper 95%	0.959	0.898	0.837	0.782	0.727	0.678
70	Lower 95%	0.910	0.819	0.741	0.669	0.606	0.548
	Median	0.932	0.853	0.780	0.715	0.655	0.599
	Upper 95%	0.954	0.888	0.826	0.765	0.708	0.656
75	Lower 95%	0.905	0.810	0.728	0.653	0.587	0.528
	Median	0.927	0.843	0.767	0.698	0.634	0.577
	Upper 95%	0.949	0.877	0.808	0.744	0.685	0.632
80	Lower 95%	0.901	0.801	0.715	0.637	0.569	0.509
	Median	0.923	0.834	0.753	0.681	0.616	0.557
	Upper 95%	0.944	0.867	0.795	0.727	0.666	0.609
85	Lower 95%	0.896	0.791	0.701	0.621	0.552	0.491
	Median	0.918	0.824	0.740	0.664	0.597	0.536
	Upper 95%	0.939	0.859	0.782	0.711	0.646	0.587
90	Lower 95%	0.891	0.782	0.689	0.608	0.536	0.471
	Median	0.913	0.815	0.727	0.649	0.579	0.517
	Upper 95%	0.935	0.849	0.768	0.695	0.629	0.568
95	Lower 95%	0.886	0.771	0.676	0.592	0.518	0.453
	Median	0.908	0.805	0.714	0.633	0.561	0.498
	Upper 95%	0.931	0.841	0.755	0.679	0.609	0.547
100	Lower 95%	0.882	0.764	0.664	0.577	0.504	0.436
	Median	0.904	0.797	0.702	0.617	0.544	0.479
	Upper 95%	0.926	0.831	0.741	0.663	0.591	0.528

Table 3. Lesser black-backed gull, demographic rate set 1, counterfactuals of population growth rate calculated between year 5 and year 30 using a matched runs method, from 5,000 density independent simulations.

Additional adult mortality	Lower 95%	Median	Upper 95%
0	1.000	1.000	1.000
5	0.999	0.996	1.002
10	0.997	0.994	1.000
15	0.996	0.993	0.999
20	0.995	0.992	0.998
25	0.994	0.991	0.997
30	0.993	0.989	0.996
35	0.991	0.988	0.994
40	0.990	0.987	0.993
45	0.989	0.986	0.992
50	0.987	0.984	0.991
55	0.986	0.983	0.990
60	0.985	0.982	0.988
65	0.984	0.981	0.987
70	0.982	0.979	0.986
75	0.981	0.978	0.985
80	0.980	0.977	0.983
85	0.979	0.975	0.982
90	0.977	0.974	0.981
95	0.976	0.973	0.980
100	0.975	0.971	0.979

Table 4. Lesser black-backed gull, demographic rate set 1, counterfactuals of population size after 5 to 30 years, estimated using a matched runs method, from 5,000 density dependent simulations.

Counterfactual of population size at 5 year intervals							
Additional adult mortality	Estimate	yr.5	yr.10	yr.15	yr.20	yr.25	yr.30
0	Lower 95%	1.000	1.000	1.000	1.000	1.000	1.000
	Median	1.000	1.000	1.000	1.000	1.000	1.000
	Upper 95%	1.000	1.000	1.000	1.000	1.000	1.000
5	Lower 95%	0.979	0.972	0.967	0.966	0.965	0.965
	Median	0.996	0.993	0.991	0.990	0.989	0.989
	Upper 95%	1.012	1.015	1.015	1.014	1.014	1.013
10	Lower 95%	0.976	0.966	0.961	0.957	0.955	0.955
	Median	0.992	0.986	0.983	0.981	0.980	0.979
	Upper 95%	1.009	1.008	1.005	1.004	1.004	1.003
15	Lower 95%	0.971	0.958	0.952	0.947	0.945	0.944
	Median	0.988	0.979	0.974	0.971	0.969	0.968
	Upper 95%	1.005	1.001	0.997	0.995	0.993	0.993
20	Lower 95%	0.968	0.952	0.942	0.938	0.935	0.934
	Median	0.984	0.972	0.966	0.961	0.959	0.957
	Upper 95%	1.001	0.995	0.989	0.985	0.983	0.982
25	Lower 95%	0.964	0.944	0.934	0.928	0.924	0.924
	Median	0.980	0.965	0.957	0.952	0.949	0.947
	Upper 95%	0.997	0.987	0.980	0.976	0.973	0.970
30	Lower 95%	0.960	0.937	0.926	0.918	0.915	0.911
	Median	0.976	0.959	0.948	0.942	0.938	0.936
	Upper 95%	0.993	0.981	0.972	0.965	0.963	0.960
35	Lower 95%	0.956	0.931	0.916	0.909	0.904	0.902
	Median	0.972	0.952	0.940	0.933	0.928	0.926
	Upper 95%	0.989	0.974	0.962	0.956	0.953	0.950
40	Lower 95%	0.952	0.924	0.908	0.899	0.892	0.889
	Median	0.969	0.945	0.931	0.923	0.918	0.914
	Upper 95%	0.985	0.966	0.954	0.947	0.943	0.939
45	Lower 95%	0.949	0.916	0.900	0.888	0.882	0.878
	Median	0.965	0.938	0.923	0.913	0.907	0.903
	Upper 95%	0.980	0.960	0.946	0.937	0.932	0.928
50	Lower 95%	0.945	0.910	0.890	0.878	0.871	0.866
	Median	0.961	0.932	0.914	0.903	0.897	0.892
	Upper 95%	0.977	0.953	0.937	0.927	0.921	0.917
55	Lower 95%	0.940	0.903	0.882	0.869	0.861	0.855
	Median	0.957	0.925	0.905	0.893	0.886	0.881
	Upper 95%	0.974	0.946	0.929	0.917	0.910	0.906
60	Lower 95%	0.936	0.896	0.872	0.858	0.849	0.844
	Median	0.953	0.917	0.897	0.883	0.875	0.870
	Upper 95%	0.969	0.940	0.920	0.908	0.899	0.896
65	Lower 95%	0.931	0.888	0.864	0.848	0.838	0.833
	Median	0.949	0.911	0.888	0.874	0.865	0.859
	Upper 95%	0.966	0.933	0.911	0.898	0.890	0.885
70	Lower 95%	0.928	0.882	0.855	0.838	0.828	0.821
	Median	0.945	0.904	0.879	0.864	0.854	0.847
	Upper 95%	0.963	0.925	0.903	0.889	0.880	0.873
75	Lower 95%	0.924	0.875	0.846	0.829	0.816	0.809
	Median	0.941	0.897	0.871	0.854	0.843	0.836

	Upper 95%	0.958	0.919	0.894	0.879	0.868	0.862
80	Lower 95%	0.921	0.868	0.836	0.818	0.805	0.798
	Median	0.937	0.890	0.862	0.844	0.832	0.825
	Upper 95%	0.954	0.912	0.885	0.869	0.857	0.851
85	Lower 95%	0.916	0.861	0.828	0.808	0.795	0.786
	Median	0.933	0.884	0.853	0.834	0.821	0.813
	Upper 95%	0.950	0.906	0.878	0.860	0.848	0.839
90	Lower 95%	0.912	0.854	0.819	0.797	0.783	0.774
	Median	0.930	0.877	0.844	0.824	0.811	0.802
	Upper 95%	0.946	0.899	0.869	0.850	0.836	0.828
95	Lower 95%	0.908	0.846	0.810	0.787	0.771	0.762
	Median	0.926	0.870	0.836	0.814	0.800	0.790
	Upper 95%	0.943	0.893	0.860	0.839	0.826	0.817
100	Lower 95%	0.904	0.840	0.801	0.777	0.760	0.749
	Median	0.922	0.863	0.827	0.804	0.789	0.779
	Upper 95%	0.940	0.886	0.852	0.829	0.815	0.806

Table 5. Lesser black-backed gull, demographic rate set 1, counterfactuals of population growth rate calculated between year 5 and year 30 using a matched runs method, from 5,000 density dependent simulations.

Additional adult mortality	Lower 95%	Median	Upper 95%
0	1.000	1.000	1.000
5	1.000	0.999	1.001
10	0.999	0.998	1.001
15	0.999	0.998	1.000
20	0.999	0.998	1.000
25	0.999	0.997	1.000
30	0.998	0.997	0.999
35	0.998	0.997	0.999
40	0.998	0.996	0.999
45	0.997	0.996	0.999
50	0.997	0.996	0.998
55	0.997	0.995	0.998
60	0.996	0.995	0.998
65	0.996	0.995	0.997
70	0.996	0.994	0.997
75	0.995	0.994	0.997
80	0.995	0.993	0.996
85	0.994	0.993	0.996
90	0.994	0.993	0.996
95	0.994	0.992	0.995
100	0.993	0.992	0.995

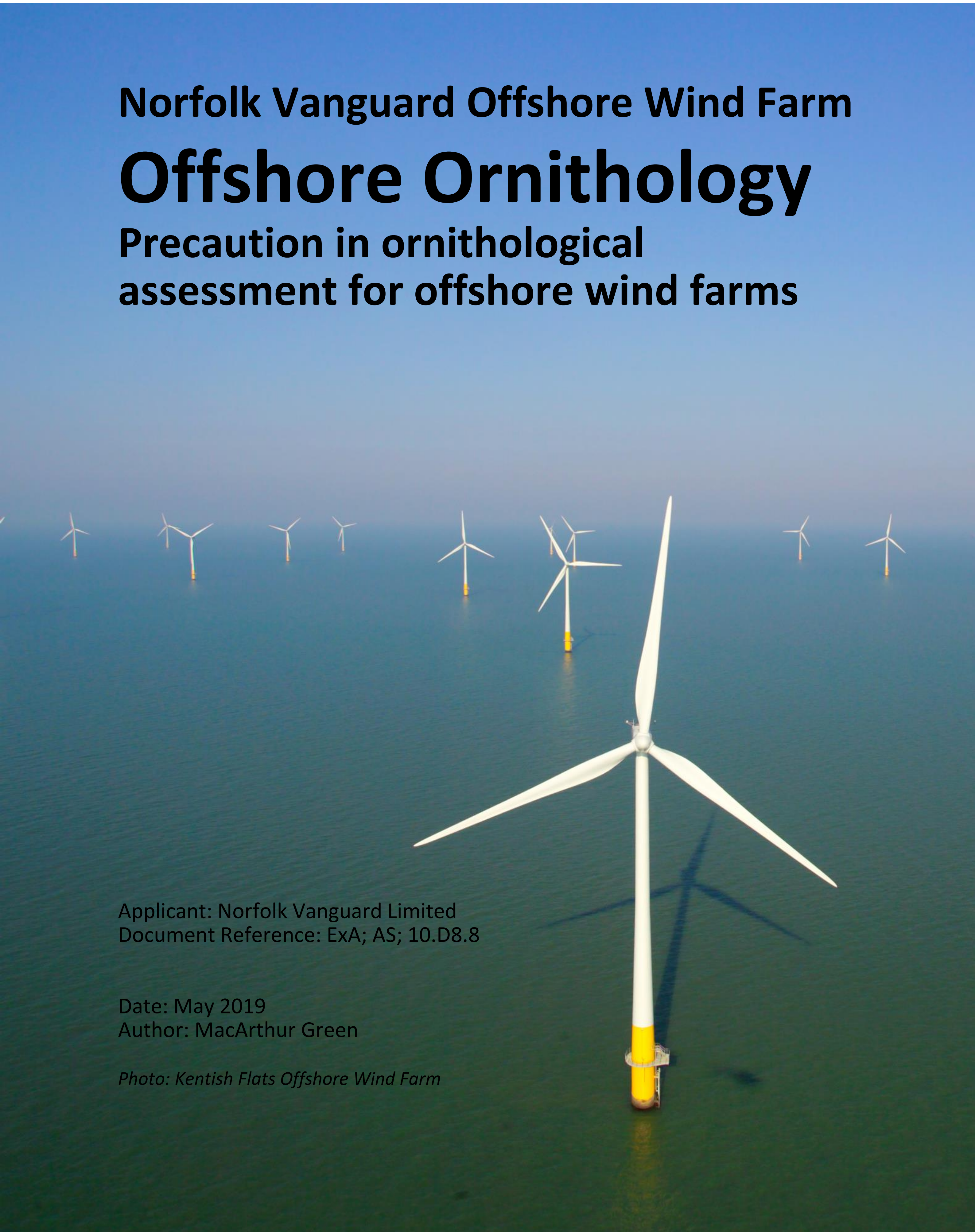
Norfolk Vanguard REP8-067: Precaution in ornithological assessment for offshore wind farms

Norfolk Vanguard Limited Reference: ExA; AS; 10.D8.8: Cited in this document as MacArthur Green 2019a

Norfolk Vanguard Offshore Wind Farm

Offshore Ornithology

Precaution in ornithological assessment for offshore wind farms



Applicant: Norfolk Vanguard Limited
Document Reference: ExA; AS; 10.D8.8

Date: May 2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
28/05/2019	01D	First draft for Norfolk Vanguard Ltd review	MT	RF	EV
29/05/2019	01F	Final for submission at Deadline 8	MT	EV	EV

EXECUTIVE SUMMARY

Ornithology impact assessments for offshore wind farms are based on extensive surveys, data analysis and modelling. The marine environment is inherently highly variable and many of the analytical methods used make allowance for the associated uncertainties, through the estimate of variance around central point estimates. It is very important that these uncertainties are given consideration in impact assessment.

However, the building block approach to impact assessment (e.g. independent estimation of the baseline population size, the magnitude of impacts and the subsequent population consequences) means that there can be a tendency to add precaution, or make precautionary assumptions, at each stage of the assessment by focussing attention on the upper limits of each component. The end result is that the final conclusion is based on considerably over-estimated impacts. This is then further compounded when individual project level impacts are added together in cumulative and in-combination assessments.

This note presents a discussion of the sources of uncertainty in ornithological impact assessments, including survey methods, data analysis, impact modelling methods and assumptions and population modelling methods. Examples from the Norfolk Vanguard assessment are used to illustrate these aspects and also to demonstrate the differences in the conclusions of an assessment based on more appropriate levels of precaution with those when multiple sources of precaution are combined without proper consideration of the probability of such unlikely outcomes.

Examples using data from the Norfolk Vanguard Offshore Wind Farm assessment are provided to highlight the scale of precaution that has been applied to the Project's impact assessment as a result of following the advice received from Natural England throughout the course of the examination. The predicted effects from combined precautionary approaches are up to 10 times greater for collision risk and up to 14 times greater for displacement risk than those obtained through the application of more appropriate methods (e.g. using mean estimates).

It is clear that there is a need to review and improve the methods for incorporating uncertainty in offshore ornithology impact assessments to replace the current approaches which greatly over-estimate impacts and produce predictions which are not only highly precautionary but also highly improbable. With respect to the Norfolk Vanguard assessment it is therefore very important to consider the extent of precaution applied to individual elements (following the methods advised by Natural England), how these individual precautions have been combined throughout the assessment to reach highly over-precautionary totals and how these have then been used by Natural England in reaching conclusions.

Table of Contents

Glossary

BDMPS	Biologically Defined Minimum Population Scale
CPGS	counterfactual of population growth rate
CPS	Counterfactual of Population Size
DCO	Development Consent Order
EIA	Environmental Impact Assessment
ES	Environmental Statement
FFC	Flamborough and Filey Coast
HRA	Habitats Regulations Assessment
MMO	Marine Management Organisation
NV	Norfolk Vanguard
PVA	Population Viability Analysis
SPA	Special Protection Area

1 INTRODUCTION

1. Offshore wind farms have the potential to impact negatively on seabirds, either as a result of the birds avoiding the turbines, which can cause birds to make longer journeys or being displaced from areas previously used for foraging (or other activities); or, if birds do not avoid the turbines, they may collide, with lethal consequences.

2. Assessing the potential magnitude of these impacts involves several steps, including data collection and analysis to estimate the baseline populations at risk; further analysis and modelling to estimate how many birds could be affected (i.e. the magnitude of potential impact); and, finally, consideration of the population consequences of the predicted impact, using methods such as population viability analysis (PVA).

3. At several stages through this process there are sources of uncertainty. These include the process of estimating seabird density and population sizes from survey data (e.g. extrapolation, boot-strapping and statistical spatial modelling); estimated values for seabird flight characteristics to be used in collision risk modelling (e.g. flight height ranges, collision avoidance rates, wingspan, etc.); and in demographic rates used in PVA models (e.g. environmental and demographic variations in survival and productivity).

4. There is a growing awareness and appreciation with the offshore wind industry that it is important to consider these (and other) uncertainties in the assessment process. However, it is also necessary for statutory agencies and regulators to apply the precautionary principle in reaching determinations of whether or not it is possible to ascertain, beyond reasonable scientific doubt and in light of the best scientific knowledge in the field, that the proposal, including any necessary mitigation measures, will not have an adverse effect on the integrity of the Special Protection Area (SPA).

5. Together, these requirements can result in a tendency for assessments to focus on impacts derived from combined upper confidence estimates and worst case scenarios. While this approach does not typically cause the impacts for an individual project to exceed levels considered acceptable (e.g. through changes in natural mortality rates), each wind farm's worst case impact predictions become the accepted figures for that project which are used in cumulative assessments for subsequent wind farm applications. This has the consequence that the total cumulative impacts can reach unacceptably high levels which, in turn, greatly exaggerate reality.

6. To take a simple example, where cumulative impact involves five offshore wind farms contributing to a total impact, if the upper 95% confidence limit is used as a precautionary estimate of collision mortality for one particular species at each of these five sites, the statistical probability of the correct value being this large is calculated by multiplying the individual probabilities of each of those estimates (i.e. a 2.5% probability at each site) together; 0.025^5 (i.e. 2.5% x 2.5% x 2.5% x 2.5% x 2.5%). This is 0.00000001, or 1 chance in 100,000,000. Clearly such a cumulative total is highly misleading, and greatly overestimates the likely cumulative impact.

7. Where a cumulative total involves 20 or more sites (such as those currently under consideration in the North Sea), the probability of the total being correct becomes too small to calculate with most pocket calculators. Yet this form of joint worst case prediction is exactly the overly precautionary approach currently being adopted by Natural England. While it is agreed that it is important to adopt a precautionary approach where there is uncertainty, it is also important to recognise that summing precaution multiple times in the same calculation quickly results in estimates reaching statistically meaningless numbers, unless based on a stochastic model that derives input values from appropriate statistical distributions for each parameter, rather than combining all extreme values (which is currently not the case).

8. It must be acknowledged that the marine environment is inherently variable and limited information may be available for certain aspects. Therefore collecting robust baseline data to inform impact assessment predictions has to strike a balance between the duration of study and the extent to which precision can be improved. Inevitably this means that some of the data used in the assessments is likely to remain uncertain. However, it is the methods by which different aspects of uncertainty are combined that can result in an assessment moving away from the application of reasonable levels of precaution to an assessment categorised by over-precaution generating statistically meaningless numbers, that should not be taken at face value in reaching conclusions on impact significance or adverse effect.

9. In the impact assessment submitted for the Norfolk Vanguard Wind Farm, attention has been drawn to the highly precautionary methods and assumptions requested by Natural England. This paper seeks to explain in further detail the nature of those precautions and how they combine to affect the results of the assessment when compared with the results obtained through the adoption of more appropriate approaches to expressing uncertainty and incorporating precaution in the assessment.

10. The following sections consider the sources of variation and uncertainty introduced in survey data and analysis methods, impact assessment methods (for collision risk and displacement), cumulative assessments based on consented rather than built

designs and population modelling. The effect of combining all these sources of precaution on the final impacts is illustrated with examples from the Norfolk Vanguard assessment.

2 SOURCES OF UNCERTAINTY

2.1 Density and abundance data

11. The recommended approach for collecting wind farm baseline data is to undertake a survey of the wind farm site and a buffer around it (e.g. 4km) each month for a minimum of two years. These surveys can be conducted from boats or planes, and are undertaken following a series of transects spaced in order to collect data across the whole site at a sampling rate of between 10% and 20% (i.e. this is the percentage of the site observed on each survey). The density of birds within the sampled area is considered to represent the density across the whole site and is therefore multiplied by the total area to obtain abundance estimates.

12. The method to estimate variance around the central value depends on the survey platform used. For digital aerial still images (as collected across Norfolk Vanguard) this was performed using a nonparametric bootstrap whereby each image was treated as the lowest sampling unit from which 1,000 random resamples were drawn. This produces a probability distribution of estimates accounting for sampling error for each survey. Because each calendar month is surveyed at least twice (note: Norfolk Vanguard East had 32 months of surveys), this process produces a probability distribution of density for each surveyed month.

13. To obtain an overall estimate for each calendar month (as input for the impact assessment models) all the survey data from each month are combined to obtain the overall spread of resampled estimates[1]. Thus, the combined 95% confidence intervals for each calendar month reflect the range from the highest to the lowest values across the two years. In other words, while the overall mean value is the mean of the means and therefore an appropriate summary across the survey data, the confidence limits are strongly influenced by the extremes from the individual years. Figure 1 illustrates this point.

14. It is apparent from the data presented in Figure 1 that the upper 95% confidence interval on density, the use of which is recommended by Natural England to represent uncertainty and which has been discussed by Natural England in relation to conclusions on impact significance (e.g. Table 1 of REP3-051), is heavily influenced in this example by the data from one of the three years, while it can be seen that the mean is more representative of all the years. While it could be argued that this is just a single example month for one species, it should be noted that the November collision estimate for gannet represents approximately half of the annual total for

[1] It should be noted that this approach was taken because the Applicant estimated collision risk using a stochastic version of the Band collision model, the results of which were presented graphically in the original application in order to explicitly present the uncertainty.

this species. It is therefore clear that by using the upper 95% confidence interval, impact assessment conclusions can be heavily weighted by a relatively small proportion of the data. Application of upper 95% confidence intervals on survey data in this manner without full consideration of the underlying distributions therefore has the potential to introduce very strong precaution from the outset of the assessment process.

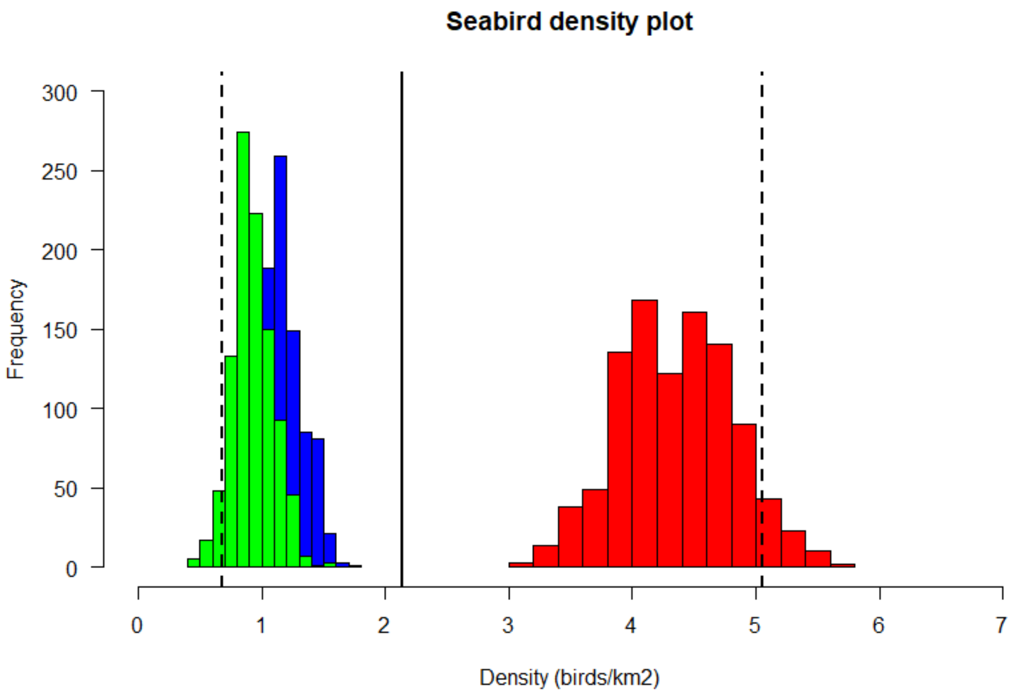


Figure 1. Gannet density data. Each coloured histogram is the bootstrapped sample for November (red: 2012, blue: 2013 and green: 2015) for gannets recorded in flight in surveys of Norfolk Vanguard East. The vertical black lines are the overall mean density (solid) and the 95% confidence intervals (dashed) estimated across all the data.

2.2 Collision risk modelling

15. Seabird density as discussed above is a key input parameter in the Band collision risk model, which is the accepted model for estimating collision risk. All the other parameters used in the modelling for Norfolk Vanguard were derived from generic datasets, most of which include estimates of uncertainty, although, for the Norfolk Vanguard collision risk assessment, only the values for monthly density, flight height, avoidance rate and nocturnal activity were adjusted in recognition of variation in these parameters. Subsequently, collision predictions were provided for the mean estimates and the upper and lower confidence values for each of these parameters separately. Natural England requested this approach on the basis that this enables uncertainty to be taken into account. However, as can be seen in Figure 1, simply

taking an upper figure and stating this is the worst case considerably over-simplifies the underlying range of data.

16. The fact that collision risk modelling should be undertaken with uncertainty captured in a more realistic manner has been accepted by Natural England, and there is a stochastic implementation of the Band collision model now available[2] (although this model was still being de-bugged during the Norfolk Vanguard assessment and examination process and was therefore not available for use). The Applicant also developed a stochastic version of the Band model in order to calculate collisions with parameter uncertainty appropriately modelled (i.e. from multiple runs of the model, with randomly generated parameter values drawn from appropriate probability distributions used in each run). However, the results from this model were not supported by Natural England, despite submissions by the Applicant which demonstrated the equivalence of the methods used with the deterministic Band model (ExA; WQApp3.3;10.D1.3). Furthermore, a key point made in the Norfolk Vanguard assessment was that interpretation of results from a stochastic model should not solely focus on the summary values, but should take full consideration of the distribution of results as provided with the original submission (ES Appendix 13.1 Annex 6). To illustrate this point, the plot of stochastic collision estimates for gannet for Norfolk Vanguard East from this submission is reproduced below.

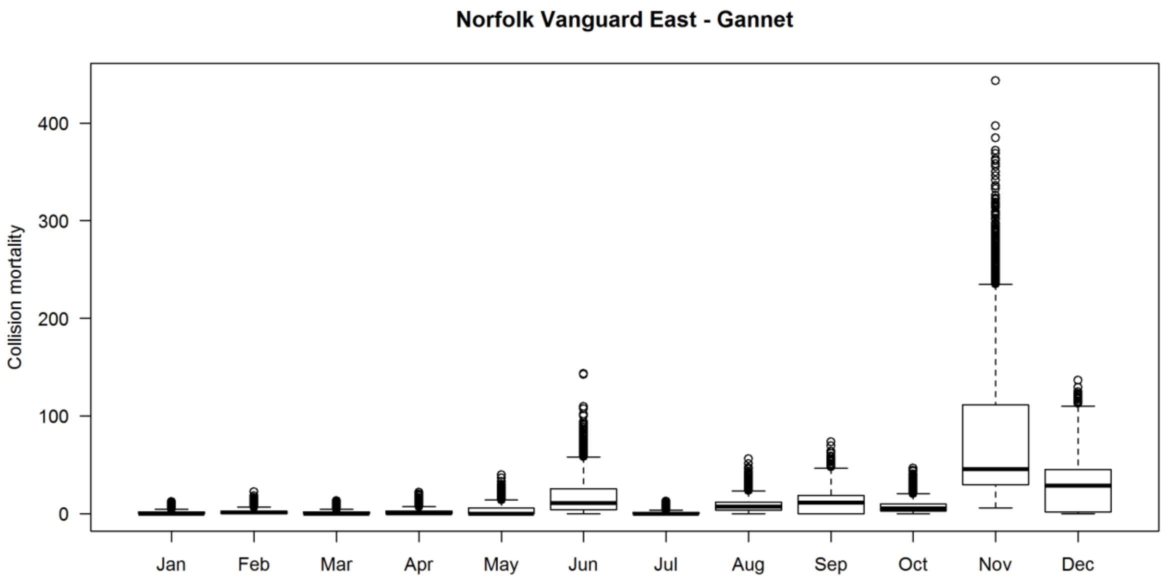


Figure 2. Norfolk Vanguard East, Gannet Option 2 collision mortality estimates calculated with stochasticity in seabird density, avoidance rate, flight height and nocturnal activity. Solid bars are the median, boxes indicate the 50% range, whiskers the 95% range and circles are outliers. (Note

[2] https://www2.gov.scot/Topics/marine/marineenergy/mre/current/StochasticCRM/fullreport

this was Figure 3 in Technical Appendix 13.1 Annex 6; it should also be noted that these collision predictions have been superseded following design revisions submitted between Deadline 6 and Deadline 8; removal of the 9M turbine, revised layout and 5m increase in draught height. However, the relative scale of monthly estimates is the same).

17. As discussed above, it is clear that the November collision predictions have a very large influence on the annual total, and the upper 95% confidence intervals for this month have a large influence on the summed annual estimate. It is also clear that the upper 95% predictions lie considerably outside the central range of predictions, and caution should therefore be taken to avoid giving this undue weight in assessing the overall impacts. The upper 95% estimate for November in Figure 2 is over 200, while the upper 50% estimate (which still retains a large degree of precaution) is less than half this value. It is clear that the November distribution of collision estimates is heavily skewed and that using the upper 95% estimate at face value, without giving consideration to the underlying data distribution, over-simplifies the situation and exaggerates the risk.

2.3 Headroom

18. Cumulative collision estimates are made up of the worst case mortality for each contributory wind farm, taken either from the relevant wind farm Environmental Statement (ES) or the Development Consent Order (DCO). Wind farm applications are submitted at an early stage in the process of the project design at which time offshore wind developers may not know the precise nature and arrangement of turbines and associated infrastructure that make up the proposed development. Assessments are therefore typically based on a project envelope approach known as a 'Rochdale Envelope' approach to impact assessment to provide flexibility for the final project design.

19. However, constructed wind farms, particularly more recent ones, rarely use the total consented number or model of turbines permitted within the consent. Technological developments mean that generating capacities can be attained with fewer, larger dimension turbines. This is important for cumulative collision estimations since collision mortalities are almost always lower for these 'as-built' developments when compared with those for the consented designs. For example, during the Norfolk Vanguard examination a 10% reduction in predicted collision was achieved with a relatively small change in minimum turbine capacity (from 9MW to 10MW). The change in turbine capacity between consented and built is often considerably greater than this, with correspondingly much larger reductions in collision risk.

20. These updates in wind farm design can be accommodated in cumulative assessments by re-calculating collision mortality for built wind farms with updated

parameters. A study undertaken to investigate the scale of reduction for key species of concern identified reductions of up to 40% between the worst case cumulative total (i.e. using consented parameters for over 30 wind farms) and the total which reflects actual built wind farms (Trinder 2017).

21. This study also presented a straightforward method for undertaking this calculation, which just uses the ratio of several key turbine parameters (consented to actual) to calculate species-specific collision adjustment rates for each wind farm. This work has been discussed with Natural England however, there has been an unwillingness to take this into account in impact assessments on the basis that unless the changes to a wind farm are legally secured there remains a potential for the consented design to be installed. While this situation is in theory apparently possible, there will be other considerations (such as constraints on the duration of construction approved by the MMO as part of the construction programme and limitations on layout approved by the MMO in the design plan) which render it so unlikely that it can be excluded in all but a very small minority of cases.

22. Given the other sources of uncertainty (and hence precaution) in collision assessments for individual wind farms, as discussed above, the use of consented, rather than built, impacts clearly adds yet another layer of precaution to an already highly precautionary process.

2.4 Displacement

23. Displacement assessments are calculated using abundance data obtained from surveys in the same way as those used in the collision assessments. These are therefore subject to the same risk of over-simplification through use of upper 95% confidence estimates without consideration of the underlying distributions. However, displacement is assessed on a seasonal rather than a monthly basis. Since the value selected to represent any particular season is the peak from the months which fall in that season, rather than the mean, the central value used in assessment is already precautionary. Natural England then request that the upper 95% confidence estimate on the peak value is used, thereby adding another layer of precaution.

24. The standard method for assessing displacement impacts is to multiply the total number of birds present (including those within a buffer of between 2 and 4 km around the site boundary, depending on the perceived sensitivity of the species, although there is very little evidence that displacement actually extends over these distances for any species) by the percentage thought likely to be displaced and by the percentage considered likely to suffer consequent mortality. The displacement percentage has been estimated at operational wind farms for several species and

there is therefore some empirical evidence available. However, it should be noted that these studies have been almost exclusively conducted at relatively old wind farms in the southern North Sea which comprise much smaller and more closely spaced turbines than those for wind farms currently in construction (or yet to be constructed), for which turbine spacing is around two times greater. There is therefore much more space between turbines within recent and planned wind farms than the study wind farms and the displacement rates are very likely to be over-estimated as a consequence (there is also emerging evidence that some species are habituating to wind turbines, Leopold and Verdaat 2018a,b).

25. A further consideration in terms of turbine layouts, which is specific to Norfolk Vanguard, is the division of turbines across the Norfolk Vanguard East (NV East) and NV West sites. For assessment, Natural England advised that the worst case should be based on the assumption that the entirety of both sites could be fully developed (and therefore could cause displacement). While Natural England acknowledged this was precautionary, it is not evident how this has been taken into consideration in their review of the assessment. The revised layout applied to the collision assessment at Deadline 6.5 (ExA; CRM; 10.D6.5.1) set limits on the proportion of turbines which would be installed, (between half and two-thirds in NV West and between half and one-third in NV East). While the areas over which the turbines may be installed across each site have not been defined, it is most probable that these would be closely related to the proportion of turbines. Thus, a realistic worst case area for displacement would in fact equate to between two-thirds and half of each site, rather than the 100% of both currently assumed.

26. The consequences of displacement are less well understood and Natural England therefore adopt precautionary values for assessment of up to 10% (i.e. 10% of displaced individuals suffer mortality as a direct result). However, the Applicant undertook its own reviews of evidence which considered all sources of information which could be used to inform this aspect and these were submitted at Deadline 1 (for red-throated diver, guillemot and razorbill; Ex; WQApp3.3;10.D1.3 and ExA; WQApp3.1;10.D1.3). These reviews concluded that realistic levels of mortality for displaced birds would be less than 1% for all the species considered. To assume a mortality rate of 1% would therefore be in keeping with the evidence and still remain precautionary.

27. It is also informative to consider the detailed individual behaviour and energetics based modelling undertaken on the potential effects of displacement on breeding seabirds (Searle et al. 2018). This is a period of the year when adults would be expected to be most at risk of negative impacts from displacement due to the reduced range over which they can forage. This study derived estimates from a

range of alternative scenarios, but typically found that adult mortality would increase by less than 1%, with chick mortality of up to around 2%. Outside the breeding season seabirds have much lower energetic requirements and have much greater freedom of movement so it seems highly unlikely that displacement during this period would have a greater effect, and much more probable that the impacts would be less.

28. In Natural England's comments on the reviews submitted for Norfolk Vanguard (Ex; WQApp3.3;10.D1.3 and ExA; WQApp3.1;10.D1.3) it was stated that their own evidence reviews, apparently based on many of the same data sources, reached different (more precautionary) conclusions (REP3-051). However, the Natural England responses focused primarily on the displacement rates, for which the differences between the Applicant and Natural England were relatively small (e.g. for red-throated diver, 100% compared with 90%, which only alters impact levels by a factor of 1.1; and, for auks, 70% compared with 50%, which alters impact levels by a factor of 1.4%). No consideration was given by Natural England to the evidence based mortality rates, for which the differences between Natural England and the Applicant are much greater (10% compared with 1%) and have much greater implications for assessment. Thus, this aspect has a much greater bearing on the impact magnitude (a factor of 10), and adds a considerable degree of precaution to the Natural England advised methods.

29. As with collision estimates, impacts of displacement that multiply several precautionary estimates (e.g. for bird density, displacement rates and consequent mortality) can result in highly improbable total displacement rates, because multiplying the upper confidence limits for three metrics results in a probability of the estimate being this large of 0.000016, or one chance in 62,500 (calculated as 2.5%, or 0.025^3). Cumulative totals based on multiple assessments for different sites further reduce this probability in a compound fashion.

30. If such calculations show that these over-estimated impacts remain tolerable, then there may be no unintended consequences with using this over-precautionary approach (although the justification for doing so remains very limited). However, the actual numbers generated cannot be taken at face value where precaution is compounded many times over and, if totals become higher than is considered acceptable, it is important to recognise the fact that these estimates are highly unrealistic. In such cases, a stochastic model based on probability distributions represents a more appropriate approach than simply combining upper confidence estimates for each parameter.

2.5 Seasonal considerations

31. Assigning impacts to individual breeding populations requires consideration of the range over which breeding birds forage; the routes taken on migration in spring and autumn (i.e. between the colony and over-wintering regions); and the ranging behaviour of immature birds which have the potential to recruit to those colonies.

32. Thaxter et al. (2012) are careful to present what they describe as 'representative' foraging ranges of seabird species, based on a wide range of study methods used at a wide range of colonies, especially in the United Kingdom. Norfolk Vanguard is located beyond the typical breeding season foraging range for most seabirds from colonies along the English coast (based on the meta-analyses in Thaxter et al. 2012). The exceptions, in terms of distance are gannet and fulmar which breed at Bempton Cliffs in Yorkshire (part of the Flamborough and Filey Coast SPA). Furthermore, while these species can cover very long distances whilst foraging, these represent the upper boundaries of such behaviour. Most trips will cover considerably shorter distances since there is strong evolutionary pressure to minimise energy expenditure and time away from the nest. If birds made multiple long trips, they would simply run out of time to provide their chicks with the numbers of feeds they require per day to survive and grow, so maximum ranges presented by Thaxter et al. (2012) represent unusual situations that could not be sustained as typical values by breeding seabirds.

33. Whilst it is still maintained that Norfolk Vanguard is of low importance during the breeding season (largely due to the distance from colonies), the site does lie in the southern North Sea in a region where large numbers of seabirds pass on migration to and from the constriction of the English Channel. For these reasons the Applicant considered that it was important to stress the presence of migrants in the impact assessment through the application of longer migration periods and the migration-free breeding seasons (as defined in Furness 2015). This was further supported in the baseline data which clearly indicated peaks of seabird abundance in spring and autumn with the lowest densities observed in the summer (i.e. when most adults will be commuting from breeding colonies to foraging areas). If large numbers of breeding birds were present at Norfolk Vanguard, then the seasonal counts would have been expected to peak in June-July when seabirds are making multiple trips to provision chicks, rather than in the spring migration period (when breeding birds tend to be attending nest sites and carrying out courtship behaviours and nest building).

34. Natural England did not agree with this approach on the basis that this was not suitably precautionary, and advised that extended breeding seasons were applied (i.e. months which fall within both migration and breeding seasons in Furness 2015

are assigned to breeding). As an example of the difference including the migration months in the breeding season, in the case of kittiwake the collision estimate for the full breeding season is 41 (March to August), while for the migration free breeding season it is 16 (May to July). The assumption that all birds present in March, April and August are breeding birds makes a large difference to the assessment but has little support from the available evidence.

35. This adds another layer of precaution in the assessment of impacts assigned to specific breeding populations because it is very probable that most, if not all, of the birds recorded in these ‘shared months’ are either late migrants heading to colonies further north or immature birds (drawn from a wide range of colonies), which are not subject to the same pressure to commence breeding and hence will be present across wider spans.

36. Further consideration of kittiwake age classes at sea is provided in ExA; AS; 10.D8.8A. This report finds that there is strong evidence to indicate that during the breeding season the density of breeding adults declines rapidly with distance offshore from colonies and is likely to be extremely low beyond 100km. While data on immature bird distributions is much more limited, all the evidence indicates that these birds will be found in greater numbers in the further offshore areas, and these are also more likely to be birds associated with Norwegian and Russian colonies.

37. This suggests that using demographically derived age structures, as is typically the case in PVA used to estimate population consequences for offshore wind farm impacts, to estimate impacts on individual age classes at wind farms located more than 100km from any particular colony will probably over-estimate the proportion of adults present and is therefore precautionary (since population growth metrics are most sensitive to changes in adult survival).

3 IMPACT CONSEQUENCES

38. The final component of the impact assessments is determining the population consequences of a predicted magnitude of impact. For all but the smallest of impacts (i.e. those which raise background mortality rates by less than 1%) this typically involves comparison of the estimated additional mortality with predictions obtained from population models.

39. The age based population models used for this purpose explicitly include demographic and environmental uncertainty, and for the most part represent one of the most robust aspects of the impact assessment process. Outputs are presented as counterfactuals of population size and growth rate (i.e. the difference between impacted and baseline projections) which have been demonstrated to be reliable and relatively insensitive to demographic uncertainty (i.e. the results are comparatively unaffected by changes in the rates of survival and productivity used).

40. However, there is a key component of population demography which Natural England does not consider should be included in the population models: density dependent regulation, with one of the cited reasons being that excluding this ensures precautionary assessment.

41. The term “density dependence” refers to the inherent regulation that occurs within populations due to competition for resources (e.g. food, mates, breeding space, etc.). While the presence of density dependence is accepted as self-evident, since without this populations would grow indefinitely, the argument for not including this in population models for seabird impact assessment has been that the mechanism for how this operates in the natural populations is insufficiently understood for it to be modelled. Furthermore, it is typically stated that the risks of including density dependence but mis-specifying the mechanism will result in completely unreliable model predictions. It is also regularly stated that density independent models, lacking any inherent means by which a population can recover once it has been reduced beyond a certain point, are therefore appropriate on the grounds of precaution (i.e. another source of precaution in the assessment process).

42. While it is undeniable that there have been very few long-term seabird population and demography studies suitable for quantifying density dependence, it does not follow that no attempt should therefore be made to include it in population models. Rather, one of the primary benefits of population modelling is that alternative methods can be investigated, the results considered against available evidence, and approaches for modelling refined in an iterative process. Furthermore, while colony specific direct measurements of density dependent regulation in action are very

rare, there is considerable evidence for density dependent regulation in seabirds, including North Sea populations.

43. The following review considers evidence for density dependence in kittiwake populations, however similar evidence is available for other UK seabirds, meaning many of the conclusions are equally applicable to other species.

44. Most demographic parameters of seabirds are likely to show some density-dependent variation (Newton 1998). Cairns (1987) pointed out that life history theory predicts that seabird breeding success will show a compensatory density-dependent response at an earlier stage of reduced food abundance and adult survival is likely to show less response until food abundance is drastically reduced. Age at first breeding may vary in a compensatory density-dependent way at an intermediate level. Empirical evidence provides some support for Cairns' predictions (Cury et al. 2011; Furness 2015). There are extensive data on breeding success of kittiwakes, showing that breeding success declines with reduction in food supply which is consistent with, but does not prove, compensatory density-dependent limitation by food supply (Frederiksen et al. 2005; Furness 2007).

45. Furness and Birkhead (1984) showed that the spatial distribution of kittiwake colonies indicated compensatory density-dependent competition for resources in the marine areas around colonies; numbers breeding at neighbouring colonies were influenced by the neighbouring kittiwake colony size.

46. Mean age of first breeding of male kittiwakes decreased from 4.59 years in 1961-70 to 3.69 in 1981-90 (Coulson 2011). The lower age of first breeding in the 1980s coincided with a much increased adult mortality, and Coulson (2011) interpreted that as evidence that competition for nest sites at the colony influenced age of first breeding, so acted in a compensatory density-dependent manner.

47. Coulson (2011) showed that the annual rate of increase in size of 46 kittiwake colonies in the UK between 1959 and 1969 was inversely related to colony size. Colonies of 1-10 pairs in 1959 increased on average by 70% up to 1969. Colonies of 10-100 pairs in 1959 increased on average by 20% up to 1969. Colonies of 100-1000 pairs in 1959 increased on average by 5%. Colonies of 1000-10,000 pairs in 1959 increased on average by 3%. This implies very strong compensatory density-dependence.

48. It is unclear, just from these changes in numbers, which particular demographic parameters were affected, but Coulson (2011) inferred that the most likely candidate is the rate of net immigration into each colony. Coulson (2011) inferred from his detailed observational studies, and from population modelling, that the

main reason for the progressive differences in growth of an individual colony is the balance between immigration and emigration of immature birds. Frederiksen et al. (2005) found that for the period 1986-2000, there was no relationship between colony size and colony growth rate, and suggested that compensatory density-dependence occurred during the expansion phase, but not necessarily at all stages of population change.

49. A compensatory density-dependent reduction in colony growth rate is also clearly evident from data on colony size over a period of decades for colonies studied in detail. Numbers at Marsden (Tyne & Wear) showed a rate of increase that progressively decreased as numbers grew (Coulson 2011, Figure 11.5). Numbers at nearby Coquet Island (Coulson 2011, Figure 11.6) show exactly the same trend with colony size. However, numbers grew rapidly at Coquet at the same time that growth had virtually ceased at Marsden (in the 1990s). This shows clearly that the rate of growth was a colony-specific feature related to local competition, and was not a consequence of region-wide variations in conditions. According to Coulson (2011) 'examination of the rates of increase of kittiwake colonies with time almost always showed the same pattern' as described above. This pattern implies compensatory density-dependence at individual colonies according to local conditions.

50. Most kittiwake colonies in the UK North Sea have declined in breeding numbers in the last few years, most strongly in the north. Decreases in numbers appear to have been greater in large colonies than in small ones, suggesting a density-dependent effect, with competition increasing most in the largest colonies as resources have declined.

51. Jovani et al. (2015) found empirical evidence from the data on the distribution of colony sizes of seabirds (including kittiwakes) in relation to breeding season foraging range for density-dependence through competition for resources around breeding colonies.

52. In conclusion, there is strong evidence, as summarised above, for compensatory density dependence acting on the kittiwake population of the UK, although exact mechanisms remain to be determined and there is some evidence to suggest that the strength of density-dependence may vary in relation to environmental conditions.

53. In acknowledgement of the uncertainty in how best to model density dependence, Trinder (2014) modelled alternative strengths of density dependence in order to determine which ones generated outputs which most closely corresponded to the evidence (the meta-analysis in Cury et al. 2011 was considered to provide the most robust guide).

54. Thus, the density dependent versions of the seabird population models to which reference has been made in the Norfolk Vanguard assessment reflect the evidence that such regulation occurs in the seabird populations of interest, and have explored alternative mechanisms for its inclusion. In contrast, the density independent versions have (for most species in most circumstances) very little support in the evidence, and from an ecological theory perspective can irrefutably be considered to be wrong, since they permit unlimited growth. Or to put this another way, density independent versions of seabird population models will always provide less reliable results than density dependent ones.

55. The consequences of using more precautionary density independent models for assessing impacts is that they will, in almost all circumstances, over-estimate the population effects of increases in mortality. This is because population growth in a density independent model is exponential (as there is nothing to limit growth). Since the baseline population projection will necessarily have a higher growth rate than the impacted one, after a typical PVA simulation duration (e.g. 30 years) the unimpacted baseline population can reach much larger sizes than the impacted one. That is, although both populations may be predicted to increase, the higher unimpacted growth rate means that it will accelerate away from the impacted one. For example, the density independent baseline prediction for the kittiwake population at Flamborough and Filey Coast SPA presented in Trinder (2014) is for an increase from the starting size of 44,000 pairs to over 150,000 after 30 years, while the 30 year population obtained with the maximum modelled level of impact was 80,000. Thus, the counterfactual of population size (CPS) for this was around 53% (80,000/150000). If the 53% figure is taken without the context of how it was obtained it would appear to be a concerning result. However, both the baseline and impacted populations have increased, and furthermore in reality neither of these predicted increases is size is likely to be feasible; Jovani et al. (2015) presented strong evidence that kittiwake colonies almost certainly can't exceed a size of around 50,000 pairs (i.e. the current size of the Flamborough and Filey Coast SPA population) before competition for resources prevents further expansion.

56. In addition to the highly unlikely density independent predictions, Natural England's approach to interpreting the PVA outputs has been to make a further assumption that the density independent CPS estimate (e.g. 53%) could apply to a stable population size (see for example the Natural England submission for Hornsea Project Three at Deadline 7[3]). Thus, the worst case density independent prediction obtained from simulations that allow considerable growth (and over estimate the differences

[3] https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010080/EN010080-001890-Natural%20England%20-%20Annex%20C%20-%20Cable%20Protection%20Advice%20Note.pdf

in population size), is applied to a population assumed to be stable at its current size (which makes an implicit assumption that the population is subject to density dependence, but without any regulatory mechanism) and then makes the assumption that this very large reduction would still apply. This mis-match of methods can be straightforwardly avoided by simply using the density dependent predictions which have been generated with an underlying assumption that the population is stable already.

57. Natural England justifies the use of density independent PVA on the grounds that they are precautionary and therefore preferable. But as has been discussed throughout this review, the effect of this is compounded by all of the preceding precautionary assumptions that are made in the estimation of the impact magnitudes, thus precautionary seabird density estimates are used to estimate precautionary impacts which are reviewed using precautionary population models. It is therefore hard not to reach a conclusion that the outcome of this is a highly over-precautionary assessment. Furthermore, these assessments are then combined with similar assessments for other wind farms to estimate cumulative effects.

58. The above compounding of precaution should be given due consideration when reviewing the Norfolk Vanguard impact assessment. The following section demonstrates the scale of the differences which occurs between estimates derived from the over precautionary approach with those obtained using more appropriate methods.

4 SYNTHESIS

59. The above discussion has presented consideration of some of the key sources of precaution which are routinely applied in seabird collision and displacement assessments for offshore wind farms in the UK. However, while it is apparent that many of these are additive to one another in terms of the final conclusions, it is not immediately apparent how different the results are when more appropriate and realistic methods are combined rather than the most precautionary. This section aims to illustrate this for collision risk and displacement.

4.1 Kittiwake collision example

60. Table 1 illustrates how the precaution in collision estimates (mean vs. upper 95%), length of breeding season (migration-free vs. full), and choice of PVA model (density dependent vs. density independent) combine to inflate impact predictions.

61. The collision estimates in Table 1 are taken from the Norfolk Vanguard assessment submitted after Deadline 7 (ExA; AS; 10.D7.5.2), and include the project revisions for removal of the 9MW turbine, the revised layout, and an increase in draught height from 22m to 27m (from Mean High Water Springs). The table provides the summed breeding season estimates (full and migration-free) and the annual totals. These have been compared with population model predictions (MacArthur Green 2018) for the Flamborough and Filey Coast SPA population, derived with and without density dependence.

Table 1. Comparison of kittiwake collision estimates at Norfolk Vanguard and PVA after 30 years.

Impact assessment stage	Migration free Breeding season		Full breeding season		Annual	
	Mean	Upper 95%	Mean	Upper 95%	Mean	Upper 95%
Collision estimate (number of individuals)	26.6	48.1	45.3	75.4	115.4	174.7
PVA Density Independent Counterfactual of Population Size (% difference)	0.85	1.54	1.45	2.41	3.69	5.54
PVA Density Independent Counterfactual of Population Growth Rate ((% difference)	0.053	0.096	0.091	0.100	0.131	0.200
PVA Density Dependent CPS ((% difference)	0.27	0.48	0.45	0.70	1.05	1.65
PVA Density Dependent CPGR ((% difference)	0.000	0.000	0.000	0.000	0.000	0.000

62. The inclusion of precaution through the use of upper 95% collision predictions and application of the extended breeding season changes the predicted collision estimate from 26.6 to 75.4 (a three-fold increase). The density independent PVA for these collisions gives a similar three-fold difference in how much smaller the

impacted population will be after 30 years (0.85% smaller compared with 2.41% smaller). If the worst case collision prediction (75.4) and worst case density independent PVA output (2.41% smaller population after 30 years) are compared with the mean collisions (26.6) and density dependent PVA output (0.27% smaller population after 30 years) it can be seen that the combined precaution amounts to an almost 10 times greater predicted effect on the SPA population.

63. However, the CPS for a density independent model is likely to exaggerate the differences between the baseline and impacted simulations since the lack of regulation permits exponential population growth. Therefore, the baseline and impact simulations can diverge by a large amount after a period of 30 years (resulting in large CPS values), although neither is likely to present realistic predictions. For density independent predictions the counterfactual of population growth rate (CPGR) is therefore likely to be more appropriate. In the example in Table 1 the density independent CPGR for the realistic breeding season collision estimate of 26.6 is 0.053%, which compares with 0.1% obtained for the precautionary collision estimate of 75.4. Thus, the CPGR for the precautionary collisions is approximately two times higher than that for the realistic collision estimate.

4.2 Guillemot displacement example

Table 2. Comparison of guillemot displacement estimates at Norfolk Vanguard East and PVA outputs after 30 years.

	Breeding season (full)			
	Monthly mean		Peak month	
	Estimate	Upper 95%	Estimate	Upper 95%
Population estimate (wind farm & 2km buffer)	1045	1930	2931	5628
Impact at 50% displaced and 1% mortality	5.2	9.7	14.7	28.1
DI PVA CPS (%)	0.18	0.33	0.50	0.96
DI PVA CPGR (%)	0.01	0.02	0.03	0.06
DD PVA CPS (%)	0.08	0.07	0.16	0.45
DD PVA CPGR (%)	0.00	0.00	0.00	0.00
Impact at 70% displaced and 10% mortality	73.2	135.1	205.2	394.0
DI PVA CPS (%)	2.49	4.59	6.88	12.82
DI PVA CPGR (%)	0.10	0.17	0.21	0.49
DD PVA CPS (%)	1.22	2.26	3.38	6.40
DD PVA CPGR (%)	0.00	0.07	0.10	0.20

64. Table 2 illustrates how the precaution in displacement estimates (mean month vs. peak month and mean estimate vs. upper 95% estimate) and choice of PVA model (density dependent vs. density independent) combine to inflate impact predictions.

65. The displacement estimates in Table 2 use the abundance estimates for Norfolk Vanguard East (Appendix 13.1, Annex 1). The table provides the breeding season estimates (full) and nonbreeding season totals. These have been compared with population model predictions (MacArthur Green 2018) for the Flamborough and Filey Coast SPA population, derived with and without density dependence.

66. It is important to note that the figures in Table 2 do not represent the actual impact apportioned to the Flamborough and Filey Coast SPA population (which varied across a small range of 0.6 to 14.7 and would therefore not generate useful variations in PVA outputs; ExA; AS; 10.D8.10). However, the figures in Table 2 provide an illustration of how realistic variations in the degree of precaution in the assessment methods generate different PVA predictions.

67. There is an almost three fold difference between the monthly mean and peak within the breeding season (1,045 vs. 2,931), and the upper 95% estimates were approximately twice the equivalent means (1,930 vs. 1,045 and 5,628 vs. 2,931). Thus, from the mean monthly abundance to the upper 95% peak estimate there is nearly six times difference in the population at risk of displacement (1,045 vs. 5,628).

68. The range of differences in abundance is mirrored in the number predicted to be impacted, and further inflated when the evidence based rates (50% displaced, 1% mortality) are compared with the precautionary Natural England rates (70% displaced, 10% mortality): all else being equal this equates to a 14x difference in predicted impact (e.g. 5.2 vs. 73.2). If the monthly mean is compared with the peak month upper 95% the difference increases to over 75x difference (5.2 vs. 394).

69. Not surprisingly, given this very wide difference between the precautionary approach advised by Natural England and the evidence based approach supported by the Applicant's literature reviews, the PVA predictions are very different. As for the number predicted to be affected, the inflation in predicted impact (in terms of density independent CPS) is:

- Due to using the peak month rather than the mean month: between 2-3x difference;
- Due to using the upper 95% estimate rather than the mean: approximately 2x difference; and,
- Due to using the precautionary rates rather than the evidence-based ones: approximately 14x difference.

70. Furthermore, the density independent model predicts impacts approximately 2x as large as those for the density dependent model.

5 CONCLUSION

71. It is clear from the results presented above that the approach currently taken to deal with uncertainty in offshore ornithology impact assessments and as recommended by Natural England, through the combination of worst case assumptions and upper confidence estimates, performed in the name of ensuring conclusions are precautionary, has in fact resulted in a process which uniformly inflates predicted impact magnitudes and subsequent conclusions on the population consequences.

72. There is therefore a need for wider discussions in the offshore wind industry to improve the understanding of how impacts, which are most appropriately defined in probabilistic terms (e.g. mean collision estimates with 95% confidence intervals), should be combined in a manner which properly captures the joint probability of realistic, but precautionary, outcomes. Simply adding the precautionary outputs from each component step, which has become common practice following the advice of statutory advisors, as detailed here, can lead to impacts which are falsely considered to represent suitable levels of precaution, when they are in fact highly improbable.

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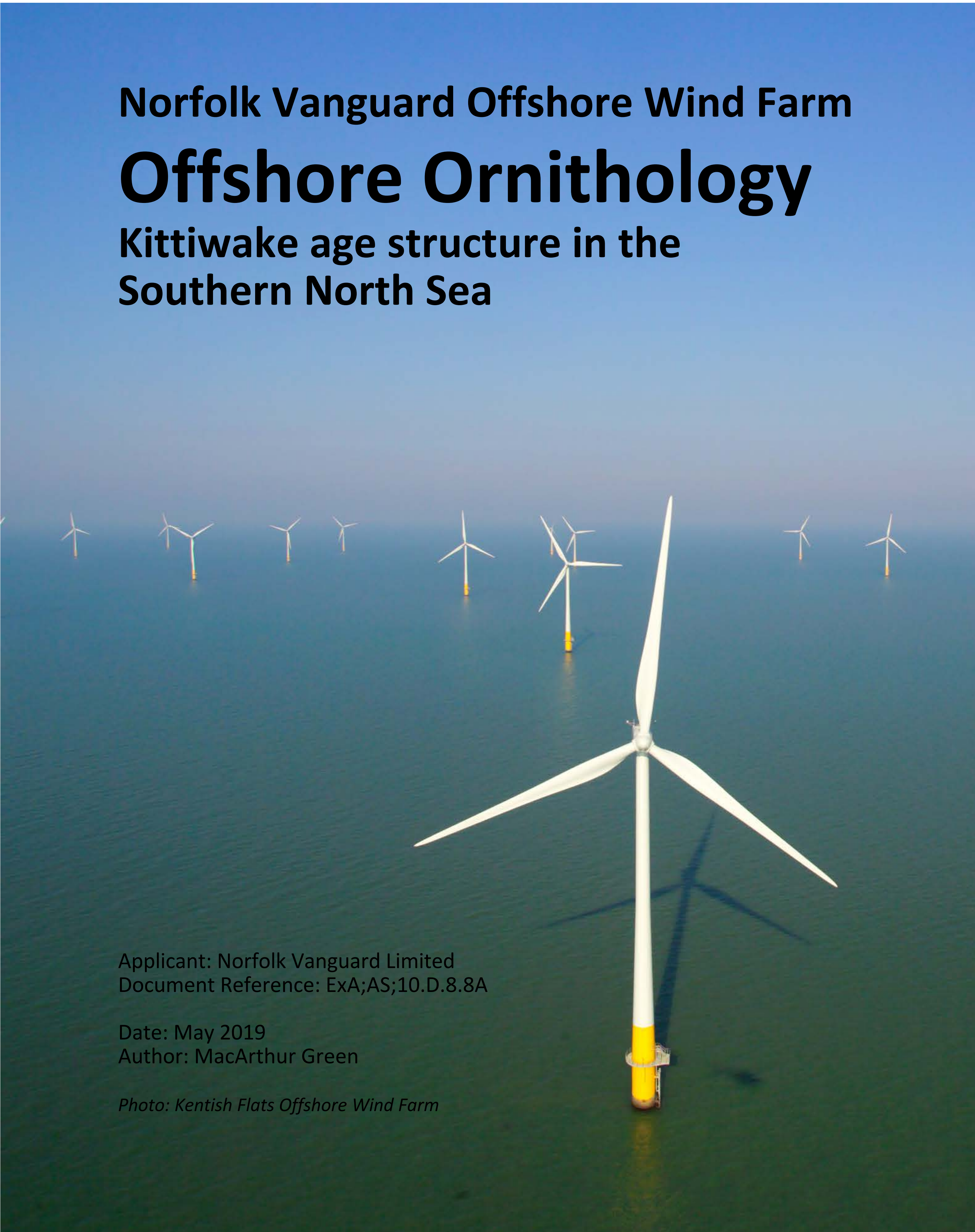
MacArthur Green

VATTENFALL

Norfolk Vanguard Offshore Wind Farm

Offshore Ornithology

Kittiwake age structure in the Southern North Sea



Applicant: Norfolk Vanguard Limited
Document Reference: ExA;AS;10.D.8.8A

Date: May 2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
30/04/2019	01D	First draft for Norfolk Vanguard Ltd review	RF	MT	EV
29/05/2019	01F	Final for submission	RF	MT	EV

EXECUTIVE SUMMARY

This note presents a review of kittiwake demographic and distribution data obtained from a variety of sources to explore the likely proportions of adult (breeding) and immature birds present at sites offshore and in relation to proximity to breeding colonies in the Southern North Sea.

The evidence strongly indicates that during the breeding season the density of breeding adults declines rapidly with distance offshore from colonies and is likely to be extremely low beyond 100km. While data on immature bird distributions is much more limited, all the evidence indicates that these birds will be found in greater numbers in the further offshore areas, and these are also more likely to be birds associated with Norwegian and Russian colonies.

This suggests that using demographically derived age structures to estimate impacts on individual age classes at wind farms located more than 100km from any particular colony will probably over-estimate the proportion of adults present and is therefore precautionary (since population growth metrics are most sensitive to changes in adult survival).

Table of Contents

1.1 How many immature kittiwakes are there in a population?

1. The best estimate of the age structure of a kittiwake population is derived from the demographic data for the population. For kittiwake colonies in east Britain (i.e. along the Scottish and English mainland North Sea coast), Horswill and Robinson (2015) recommend the use of the following demographic data: age of recruitment 4 years old, juvenile (0-1 year) survival 0.790, adult (2+ years) survival 0.854, incidence of missed breeding 0.180-0.208 and productivity 0.819 chicks per nest. With the exception of juvenile survival, which is based on a single study from several decades ago, all the other demographic data were considered by Horswill and Robinson (2015) to be of especially high quality and reliability, and appropriate for modelling east coast UK populations.

2. Productivity is higher at colonies in the Southern North Sea than at colonies in the northern North Sea (Cook and Robinson 2010; Horswill and Robinson 2015), and productivity is the most important factor influencing the rate of increase or decrease of kittiwake colonies (Coulson 2017). Kittiwakes in Britain need to produce about 0.8 chicks per nest in order to maintain a stable population size (Coulson 2017). Based on the demographic data, a colony of 100 breeding pairs of kittiwakes on the east coast of England would fledge 82 chicks (100 x 0.819 chicks per nest). From the breeding population, 29 adults would die (200 x (1-0.854)), and would be replaced by immature birds surviving from the cohort of 82 chicks fledged four years previously. In addition, with about 20% of adult age birds taking a year off breeding, the colony of 100 breeding pairs would have another 40 breeding-age birds associated with it but not breeding in a particular year. It can be anticipated that 6 of these birds will die each year (40 x (1-0.854)). The demographic data predict that there will be 82 fledglings, 63 1-year olds (82 x 0.790), 50 2-year olds (63 x 0.790), 43 3-year olds (50 x 0.854), and 37 4-year olds (43 x 0.854), with the 4-year olds replacing the mortality of 29+6=35 adults that die. These demographic data come close to matching the predicted stability of the kittiwake population in east England with observed productivity and other demographic parameters, further reinforcing the evidence that the demographic data are appropriate.

3. Based on these demographic data, an east of England kittiwake colony of 100 breeding pairs will have an age structure of 82 juveniles, 63 1-year olds, 50 2-year olds, 43 3-year olds, and 240 breeding-age birds, giving a total population of 478 birds. Therefore, breeding age birds represent 50.2% of this population while immatures represent 49.8% of the population. Based on a similar analysis of a model population of kittiwakes, Furness (2015) estimated that there are 0.88 immatures per adult kittiwake in the UK population (i.e. adults comprised 53% of the total population).

4. Almost half of the kittiwake population is immatures rather than breeding age birds. Furthermore, since about 20% of breeding age birds do not breed in any particular year (Horswill and Robinson 2015), the 200 breeding adults at the 100 nests in the colony at which eggs were laid have 278 nonbreeding birds (immatures plus nonbreeding adults) associated with them. So there are 1.39 nonbreeding birds for each breeding adult in the kittiwake population.

5. Since counts of kittiwake colony sizes are based on counts of 'apparently occupied nests' the census unit is somewhere between the 100 breeding pairs and the 120 potential pairs with nonbreeding birds included (since some pairs of kittiwakes will build a nest but not lay eggs, so are nonbreeding birds that would be included in the population census of apparently occupied nests). There is, therefore, some uncertainty about the ratio of all kittiwakes in the population to numbers of nests counted when colonies are censused.

6. **Conclusion: it is clear that there are about as many immature birds in a kittiwake population as there are breeding birds.**

1.2 At what latitudes are adult and immature kittiwakes during the nonbreeding season?

7. Movements of breeding adult kittiwakes have been investigated by deployment of geolocator tags (Frederiksen et al. 2012). Many breeding adult kittiwakes cross the Atlantic in late summer to spend part of the nonbreeding season off Newfoundland and Greenland. Birds from colonies at higher latitudes tend to remain during the nonbreeding season at higher latitudes than birds from more southerly breeding areas, so the distribution of breeding adults in the nonbreeding season is somewhat segregated by latitude (Frederiksen et al. 2012). However, Frederiksen et al. (2012) present electronic supplementary material to their paper estimating that 255,261 adult kittiwakes were present in the entire North Sea (not just the UK portion) in December, with 114,195 of these being birds from North Sea colonies, 102,671 from Barents Sea colonies, 24,071 from Norwegian Sea colonies, and 14,324 from Celtic Shelf colonies. Therefore, numbers of breeding adults from Barents Sea colonies roughly equalled numbers of breeding adults from North Sea colonies in the North Sea in mid-winter.

8. Less is known about the at-sea distribution of immature kittiwakes, but despite the problem of biases in ring recovery data (Coulson 1966) it is evident that immature kittiwakes generally tend to be distributed further south than breeding adults from the same population (Coulson 1966; Wernham et al. 2002; Coulson 2011). This means that the numbers of immature kittiwakes from the Barents Sea population that are present in the North Sea in winter are likely to be considerably larger than

the numbers of breeding adults from that population wintering in the North Sea (because many adults from Barents Sea colonies do not come as far south as the North Sea whereas more of the immatures from that population do come as far south as the North Sea). Conversely, immature kittiwakes from UK populations are likely to be less numerous than breeding adults from North Sea colonies in the North Sea in winter, since more immatures from UK colonies are likely to be further south (Coulson 1966, 2011).

9. The timing of spring migration of UK kittiwakes and those from higher latitudes is very different, and this will influence the proportions of birds at-sea that are from these different populations during the spring months. The first UK kittiwakes normally return to colonies around the Southern North Sea in January and February. During January and February, kittiwakes are only present at North Sea colonies intermittently, though progressively longer through the day as the date progresses. By March, Southern North Sea kittiwake colonies are occupied throughout the day, with about 20% of nest sites occupied by pairs (Coulson 2011). In Shetland, arrival is mainly in February rather than January (Pennington et al. 2004), apparently about a month later than at colonies in the Southern North Sea. In contrast, at high latitude locations, such as Svalbard, the first kittiwakes to return usually do not arrive until April or May, with the latest recorded first arrival being on 31 May (Løvenskiold 1963). After these first birds, the main arrival back at colonies there occurs in late April or early May in most years (Løvenskiold 1963). Belopol'skii (1961) reported the mean date of the first return of kittiwakes as 19 April to Spitsbergen, 21 April to Franz Josef Land, and 29 April to Novaya Zemlya. Although Løvenskiold (1963) provides the most detailed data on arrival time, more recent observations (Anker-Nilssen et al. 2000) show that the 1.8 million adults breeding at colonies in the Barents Sea mostly return in April, much the same timing as previously reported by Løvenskiold (1963). Therefore, many of these breeding birds from high latitude colonies will still be at-sea in the Southern North Sea while UK kittiwakes are predominantly already standing on their nest sites on the cliffs at UK colonies.

10. Furness (2015) suggested that the UK North Sea waters BDMPS may hold about 830,000 kittiwakes in autumn (August to December), with about 430,000 of these originating from the UK, while the spring BDMPS (January to April) may hold about 630,000 birds, with about 390,000 of these originating from the UK, although the difference in timing of spring migrations of UK and high latitude populations may result in at-sea proportions in spring being much more weighted towards high latitude populations.

11. **Conclusion: During the non-breeding season, many kittiwakes in the North Sea are likely to be from the Barents Sea. Birds from North Sea colonies probably represent**

only about 50-60% of those present in autumn, and may be less than 50% of the birds at-sea in spring, since many UK adults will be attending nest sites rather than being at-sea.

1.3 At what latitudes are adult and immature kittiwakes during the breeding season?

12. Breeding adults and those nonbreeding adults that are attending the colony spend the breeding season at approximately the latitude of their colony. Some immature kittiwakes also attend the colony during the breeding season, without being represented in the census of 'apparently occupied nests'. Coulson (1966) estimated that possibly half to three-quarters of the 1-year old and 2-year old kittiwakes were within 500 miles of their natal colony in summer, often spending time resting on shorelines in that general area. However, such quantitative estimates are very difficult given the strong bias in ring recoveries, with under-representation of offshore areas likely to overestimate numbers returning to coasts (Coulson 2011). Some, possibly many, of the younger immature kittiwakes remain at-sea through the summer, away from their colony (Coulson 2011).

13. It is understood from ring recovery data that many of these young immature kittiwakes spend the summer at lower latitudes than their area of birth (Coulson 1966; Wernham et al. 2002; Coulson 2011). In particular, many young kittiwakes from British colonies spend the summer in waters off Spain and France at 40-50°N (a region not normally visited by adult kittiwakes from the UK and where the local breeding population is extremely small; Coulson 2011), whereas many young kittiwakes from colonies at high latitudes such as Russia and Norway spend the summer at about 50-60oN (Coulson 1966). There is, therefore, a spatial separation between the at-sea distributions of immature kittiwakes from different latitudes, as well as a tendency for immatures to be found further south than adults.

14. Very few kittiwakes breed south of the English Channel; 5 pairs in Portugal, 200 pairs in Spain, a few thousand pairs in Atlantic France (Mitchell et al. 2004). By comparison, Seabird2000 found 370,000 pairs in the British Isles in 1998-2002, whereas the majority of the breeding population is further north (Iceland, Faroe Islands, Norway, and Russia were estimated to hold about 2 million pairs; Mitchell et al. 2004). Therefore, there are very large numbers of immature kittiwakes from these higher latitude colonies, and the evidence is that many of these immatures spend the summer at lower latitudes than their natal colonies (Wernham et al. 2002). Some of these immatures will be present in the North Sea during the summer. Whereas UK breeding kittiwakes spend about half of their time at their nest site and half at-sea, kittiwakes from high latitude populations that summer in the North Sea are thought to remain continuously at-sea, and not to come onto land. Therefore,

the ratio of birds from high latitude colonies to adults from UK colonies will be further altered by the fact that the UK adult kittiwakes spend half of their time at their nest site.

15. In addition, the European Seabirds at Sea (ESAS) data (counts of seabirds at sea) indicate that there are 850,000 (range 600,000 to 1.1 million) kittiwakes in the North Sea during summer (Campuysen et al. 1995; WWT Consulting and MacArthur Green 2013; Furness 2015; NERC MERP data in prep.). Since there were about 300,000 kittiwake apparently occupied nests at UK North Sea colonies (including Orkney and Shetland in this total) in 1985-87 and also in 1998-2002 (Mitchell et al. 2004) during the decades when most of the ESAS data were collected, and since through much of the breeding season slightly fewer than half of the birds from these nests would be at sea at any particular time during the day and slightly more than half attending the nest (Coulson 2011), only about 300,000 of the 850,000 (range 600,000 to 1.1 million) kittiwakes in the North Sea in summer are likely to be breeding adults from UK colonies. The remainder (mean estimate 65%, range 50% to 73%) are likely to be immatures from the UK population and from higher latitude populations and nonbreeders from the UK and higher latitude populations.

16. **Conclusion: During summer, North Sea waters hold large numbers of breeding adult kittiwakes, but also hold large numbers of immature and nonbreeding adult kittiwakes, including large but uncertain numbers from the Barents Sea population. The proportions in these different categories are unclear, as they cannot be quantified from available data, but it seems highly likely that breeding adults from UK colonies represent less than 50% of the kittiwakes present over North Sea waters in summer.**

1.4 Within the Southern North Sea, whereabouts at-sea are breeding and immature kittiwakes during the breeding season?

17. Breeding kittiwakes are central-place foragers, based at their nest site. From there, they travel out to sea to forage. Theory predicts that birds should forage as close to the colony as they can, to minimize time and energy costs of commuting flight from the nest to the feeding area (Cairns 1987, 1992). Tracking data from breeding adult kittiwakes support that prediction; densities of breeding adult kittiwakes foraging at-sea tend to decline with distance from a colony (Wakefield et al. 2017), and tend to decline faster around small colonies than around large colonies; i.e. foraging distances show density-dependence, with larger foraging ranges from larger colonies (Wakefield et al. 2017). Foraging ranges also tend to be longer from colonies where food supply has declined (Bolton and Owen 2012), and productivity is lower at these colonies (Miles 2012; Coulson 2017), further supporting the interpretation of density-dependent competition for food around colonies during the breeding

season. There is also clear evidence for density-dependence in the rate of growth of kittiwake colonies: larger colonies tend to grow more slowly than smaller colonies (Coulson 1983). In addition, the spatial distribution of kittiwake colonies also indicates density-dependent competition for food; colonies near to large colonies tend to be small, and further away, than where colony sizes are smaller (Furness and Birkhead 1984), and birds travel further to forage from colonies with limited access to the sea (Wakefield et al. 2017).

18. Since there are relatively few kittiwake colonies in the Southern North Sea (Mitchell et al. 2004), the density of breeding adult kittiwakes at-sea in UK Southern North Sea waters can be predicted from the locations and sizes of those few colonies. The prediction would be that breeding adult kittiwake density at-sea would decline from a peak immediately beside each colony to close to zero at distances exceeding the maximum foraging range of breeding adult kittiwakes. Maximum foraging range varies among studies. Daunt et al. (2002) found that breeding adults from the Isle of May travelled less than 73 km from the colony. Thaxter et al. (2012) found from eight studies that maximum foraging range averaged 60 km, with a mean range of 24.8 km across these eight studies. Subsequent tracking has found higher maximum ranges for kittiwakes from Flamborough and Filey, the largest colony in the North Sea (Wischnewski et al. 2018) and from some colonies in Orkney and Shetland, where breeding success was zero or close to zero due to food shortage in the region (Bolton and Owen 2012). Wischnewski et al. (2018) reported a mean foraging range of breeding adult kittiwakes tracked from Flamborough and Filey Coast SPA of 89 km, and a maximum range of 324 km. The tracking data suggest that kittiwake density at-sea would be likely to decline considerably over the first 50 km from each colony, and would decline to close to zero beyond 100 km from most kittiwake colonies. The exceptionally long foraging trips reported from Flamborough and Filey Coast SPA suggest that breeding adults from that colony may extend further from the colony than is normally the case elsewhere, but even in this case, density of breeding adults will decrease considerably with distance from the colony. There may also be aspects of the tracking work at Flamborough and Filey Coast SPA that bias the results, as the very small number of tracked birds were primarily from nests at the edge of the colony, so may represent the lowest quality adults at the colony (Coulson 2011). If those birds are less competitive than most adults that may explain why their foraging trips are exceptionally distant (to avoid competition with higher quality birds that outcompete them over waters closer to the colony).

19. How do these predictions match empirical evidence? Empirical data on at-sea density of kittiwakes in the Southern North Sea are available from the European Seabirds at-sea (ESAS) database (WWT Consulting and MacArthur Green 2013; Bradbury et al. 2014). These data show a very different pattern from that predicted

just for breeding adults. Densities of kittiwakes at-sea during summer show very little decline with distance from east coast colonies (WWT Consulting and MacArthur Green 2013, Figure 21). Densities of kittiwakes at-sea in summer remained around 1-4 birds per km2 from the Northumberland to Lincolnshire coast to as far as 300 km offshore. Since tracking data show much higher densities of breeding adult kittiwakes close to colonies than further away, the ESAS data suggest that a high proportion of the kittiwakes at-sea further offshore from the coast are immatures or non-breeders rather than breeding adults, with these immatures and non-breeders showing a very different spatial distribution from that of the breeding adults. This is exactly what theory would predict: immatures are likely to be less competitive than breeding adults because they are less experienced, while nonbreeders are likely to be less competitive than breeding adults (they have presumably chosen not to breed due to being in poorer body condition, which is either a consequence of the individual being less competitive or will likely cause it to be less competitive). Since kittiwakes are subject to density-dependent competition for food at-sea (Wakefield et al. 2017), it is to be expected that the less competitive immatures and nonbreeders will avoid areas with high numbers of breeding adults (such as close to colonies) and will distribute themselves across marine areas distant from colonies where intra-specific competition is lower. Comparison of the ESAS data and tracking data from breeding adults strongly supports this prediction based on theory. Exactly this sort of spatial segregation of adults and immatures in relation to colony location has recently been demonstrated from aerial survey observations of gannets in the English Channel and Bay of Biscay (Pettex et al. 2019). That study found that during the breeding season, adult gannets were constrained by central place foraging whereas immatures filled in the habitat in areas more distant from colonies, showing very little overlap between these age classes. This spatial segregation of age classes, predicted by Wakefield et al. (2017) and demonstrated by Pettex et al. (2019) is likely to apply to all seabird species during the breeding season, and possibly during the nonbreeding period for those species where adults tend to remain closer to their breeding area than immature birds do.

20. **Conclusion: Theory, and empirical evidence from the ESAS data compared to evidence from tracking of breeding adult kittiwakes, suggest that the proportion of foraging immature and nonbreeding kittiwakes increases from close to zero immediately adjacent to colonies up to close to 100% at distances more than 100 km from most kittiwake colonies.**

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Norfolk Vanguard REP8-104 Norfolk Vanguard Natural England's Comments on Offshore Ornithology Final D8

Natural England reference EN010079 280952: Cited in this document as Natural England (2019b)

THE PLANNING ACT 2008

THE INFRASTRUCTURE PLANNING (EXAMINATION PROCEDURE) RULES 2010

NORFOLK VANGUARD OFFSHORE WIND FARM

Planning Inspectorate Reference: EN010079

Natural England's Comments on Norfolk Vanguard Ltd. Deadline 7 and Deadline 7.5 submissions in relation to Offshore Ornithology Related Matters

30 May 2019

This document is a technical document submitted into the Norfolk Vanguard Examination to provide scientific justification for Natural England's advice provided on the significance of the potential impacts on designated sites features, as summarised within each section. Our advice is based on best available evidence at the time of writing and is subject to change in the future (likely to be outside of this examination process) should further evidence be presented.

Table of Contents

1. Summary

1.1. The Norfolk Vanguard alone and cumulative at EIA scale and alone and in-combination at HRA predicted collision risk impact figures have been updated by the Applicant since Natural England's Deadline 7 response in REP7-075, and indeed since the Applicant's own Deadline 7 submissions. The figures have been updated following revised collision risk modelling (CRM) by the Applicant to account for increased draught height for the Vanguard project (as presented in the Deadline 7.5 'Deterministic Collision Risk Modelling for revised layout scenarios and increased draught height' document, AS-049). We have therefore based our comments below on the evidence presented by the Applicant in AS-049 and the 'Cumulative and In-combination collision risk assessment update', AS-048.

1.2. We welcome the Applicant has now considered this mitigation measure, as Natural England has recommended that increased draught height be considered since our advice on the Preliminary Environmental Information Report (PEIR). This has further reduced the level of impacts predicted by the Applicant, following the previous refinement of the 'worst case scenario' at Deadline 6.5 (AS-043). Nevertheless, the project continues to make a meaningful contribution to cumulative and in-combination effects on several seabirds at both the EIA scale and with respect to qualifying features of seabird colony SPAs through collision mortality, particularly with respect to North Sea populations of great black-backed gull, gannet and kittiwake, Flamborough and Filey Coast SPA kittiwake and gannet, and Alde-Ore Estuary SPA lesser black-backed gull (see Table 1). Detailed advice on these and other receptors follows.

Table 1 Summary of conclusions for assessment of collision risk from Vanguard alone and cumulatively / in-combination with other plans and projects for relevant species following Applicant's updated collision risk assessment in AS-048 to account for revised layout scenarios and increased draught height

EIA species	**Collision risk alone**	**Collision risk cumulative**
Gannet	Not significant (negligible to minor adverse)*	Significant (moderate adverse)*
Kittiwake	Not significant (negligible to minor adverse)	Significant (moderate adverse)
Lesser black-backed gull	Not significant (negligible to minor adverse)	Not significant (minor adverse)
Herring gull	Not significant (negligible to minor adverse)	Not significant (moderate adverse)
Great black-backed gull	Not significant (negligible to minor adverse)	Significant (moderate adverse)
Little gull	Not significant (negligible to minor adverse)	Unable to reach a conclusion due to missing sites in the cumulative assessment
HRA species & site	**Collision risk alone**	**Collision risk in-combination**
Gannet: Flamborough & Filey Coast SPA	No adverse effect on site integrity (AEOI)*	No AEOI excl. H3* AEOI incl. H3*
Kittiwake: Flamborough & Filey Coast SPA	No AEOI	AEOI excl. and incl. H3
Lesser black-backed gull: Alde-Ore Estuary SPA	No AEOI	AEOI excl. H3 (no collisions apportioned from H3)
Little gull: Greater Wash	No AEOI	Unable to reach a conclusion due to missing sites in the in-combination assessment

* Gannet considered for collision risk plus displacement for assessments of Vanguard alone and cumulatively/in-combination with other plans and projects

1.3. Natural England has previously provided regulators with our advice regarding our concerns about predicted level of cumulative impacts on North Sea seabirds, e.g. EIA great black-backed gull at East Anglia 3, Flamborough and Filey Coast (FFC) SPA kittiwake at Hornsea 2, which were subsequently consented. These concerns have intensified during the recent three offshore wind farm (OWF) examinations (Hornsea 3, Norfolk Vanguard, Thanet Extension), and given four further OWF NSIPs are due to be submitted to PINS in the next twelve months (Norfolk Boreas, East Anglia One North, East Anglia Two, Hornsea Four), Natural England considers that without major project-level mitigation being applied to all relevant projects coming forward, there is a significant risk of large-scale impacts on seabird populations.

1.4. Natural England therefore recommends that for all relevant future projects located in the North Sea, raising turbine draught height should be considered as standard mitigation practice, and that where appropriate relevant proposals should include this measure in order to minimise their contributions to the cumulative/in-combination collision totals by as much as is possible.

1.5. No further updates have been made by the Applicant with regard to auk displacement assessments since the updated assessment in the Deadline 6 documents (REP6-021. Therefore, our position regarding auk displacement alone and cumulatively/in-combination remains as set out in our Deadline 7 response (REP7-075). However, we understand from discussions with the Applicant that they intend to submit updated auk cumulative and in-combination assessments at Deadline 8. Natural England will endeavour to respond to these at Deadline 9.

2. Environmental Impact Assessment (EIA)

2.1. EIA impacts from collision risk from Vanguard alone

2.1.1. The Vanguard alone predicted collision risk impact figures have been updated by the Applicant since our Deadline 7 response in REP7-075. The figures have been updated following revised collision risk modelling (CRM) by the Applicant to account for increased draft height (as presented in AS-049).

2.1.2. Natural England has evaluated the CRM outputs presented by the Applicant in the Deadline 7.5 'Deterministic Collision Risk Modelling for revised layout scenarios and increased draft height' document, AS-049 for each of the six key species considered to be at risk of collision impacts at an EIA scale: gannet, kittiwake, lesser black-backed gull (LBBG), herring gull, great black-backed gull (GBBG) and little gull.

2.1.3. With regard to the figures presented in the Deadline 7.5 CRM for the updated layout scenarios and increased draft height (AS-049), we understand that the input parameters used, including the mean bird densities and upper and lower 95% Confidence Intervals (CIs) of this, are the same as those presented in Appendix 1 of REP6-019 (with the exception of the turbine revs per minute and turbine hub height). We have therefore reviewed the CRM outputs for the revised layout scenarios and increased draft height using the updated figures for turbine rpm, hub height and turbine numbers in each of Vanguard West and East, but retaining the other parameters, including the mean bird densities and associated CIs. We agree with the predicted figures given by the Applicant in Table 2 of AS-049 for the central (based on mean density) for the revised CRM, but we do not get the same ranges of figures based on the 95% CIs of the density data.

2.1.4. Based on the updated predictions for the WCS turbine layout option (namely 1/2 of the turbines in Vanguard West and 1/2 in Vanguard East for gannet, kittiwake, herring gull, GBBG and little gull and 2/3 of the turbines in Vanguard West and 1/3 in Vanguard East for LBBG) with the increased draft height, we note that based on the Natural England calculated ranges presented in Table 2 below, all the central CRM predictions (i.e. using mean density, mean avoidance rate, maximum likelihood flight height data and the standard nocturnal activity rates) equate to less than 1% baseline mortality of the largest BDMPS and biogeographic populations for all of the five key species. This is also the case for the upper 95% confidence intervals of the bird density for all species.

Table 2 Percentage of baseline mortality for CRM for vanguard alone at EIA scale, using average across all age class mortality rates, as used by the Applicant

	CRM prediction, Vanguard alone, Deadline 7.5,Table 2 AS-049*	**Largest BDMPS (North Sea) individuals, Furness (2015)**	**% baseline mortality largest BDMPS Deadline 7.5, AS-049***	**Biogeographic population individuals (Furness 2015)**	**% baseline mortality biogeographic Deadline 7.5, AS-049***
Gannet	66 (12-161)	456,298	0.08 (0.01-0.18)	1,180,000	0.03 (0.01-0.07)
Kittiwake	115 (12-300)	839,456**	0.09 (0.01-0.23)	5,100,000	0.01 (0.002-0.04)
LBBG	23 (1-66)	209,007	0.09 (0.004-0.25)	864,000	0.02 (0.001-0.06)
Herring gull	14 (0-52)	466,511	0.02 (0-0.06)	1,098,000	0.01 (0-0.03)
GBBG	47 (1-155)	91,399	0.28 (0.001-0.92)	235,000	0.11 (0.002-0.36)
Little gull	5 (0-12)	10,000***	0.25 (0-0.6)	75,000****	0.03 (0-0.08)

* Note discrepancies in figures calculated by Applicant for the range based on 95% CIs of bird density and those calculated by Natural England. The figures calculated by Natural England are presented above
** Population estimate for all UK colonies within North Sea BDMPS scale (from Furness 2015)
*** Precautionary estimate based on the surveys conducted across the Greater Wash Area of Search and analysis of those data in Natural England & JNCC (2016), as used by Applicant
**** Little gull population with connectivity to the southern North Sea was estimated to be up to 75,000 (Stienen et al. 2007), as used by Applicant

2.1.5. **<u>Therefore, based on these revised figures we conclude that the collision risk from Norfolk Vanguard alone would have no significant impact at the EIA scale for all species.</u>**

2.2. EIA cumulative impacts with other plans and projects

a. <u>General comments</u>

2.2.1. In AS-048, the Applicant refers to projects in the cumulative and in-combination assessments that have been built out to a lower capacity than that consented as a source of precaution within the assessments. As we have stated previously (see our Deadline 2 response, REP2-038, to the Applicant's Section 51 advice response AS-006) Natural England acknowledges that this is an important issue with regard to cumulative/in-combination CRM predictions and assessments. However, without a legally secured reduction in the consented Rochdale envelope, and a re-run CRM with the final design parameters, cumulative assessments should be based on consented parameters. We note that East Anglia 1 is the only project to date to meet these tests.

2.2.2. The Applicant also refers to nocturnal activity factors used in the assessments as being overestimates. As we have noted previously (in our Relevant and Written Representations, RR106, REP1-088), we recognise that from recent evidence presented e.g. by MacArthur Green (2015) and Furness et al. (2018), nocturnal activity levels relative to daytime levels for some species may be lower than the levels that equate to the nocturnal activity factors currently used in CRM. However, our position regarding nocturnal activity rates/factors has been set out in RR-106, REP1-088, REP2-038 and our position remains unchanged, which includes that offshore survey periods will have missed peak activity around dawn and dusk and therefore it is not appropriate to apply 'empirically derived' nocturnal activity rates from tracking studies to offshore survey recorded results. Additionally, as we have previously noted in REP2-038, it is not appropriate to simply adjust the CRM figures for the other OWFs included in the cumulative assessments to account for a change in nocturnal activity rate without re-running the CRM, as the modelling calculates the reduction in activity at night through the interaction of nocturnal activity and the latitude of the specific wind farm. Therefore this is a calculation specific to that wind farm and hence a re-run of the model is required.

2.2.3. We note that the figures the Vanguard Applicant has used in the cumulative and in-combination assessments in AS-048 are in some cases (e.g. gannet, kittiwake) different to those used by Natural England in our Deadline 7 response at Hornsea 3 (Natural England 2019). The Hornsea 3 figures we used in Natural England (2019) were essentially the Hornsea 3 Applicant's data but with our preferred parameters, and were presented for illustrative purposes only. <u>We have used the same figures for Hornsea 3 in this response as we have used in the Hornsea 3 Deadline 7 response to ensure consistency across the projects (Natural England 2019)</u>. Our advice remains that there is still considerable uncertainty around the Hornsea 3 cumulative contribution due to the lack of a full baseline dataset, hence our suggestion that NV present figures with and without Hornsea 3.We also note that the cumulative/in-combination total figures that the Vanguard Applicant presents in their updated assessments in AS-048 are different to those presented by Natural England in our Deadline 7 response at the Hornsea 3 examination (Natural England 2019). This is partly due to the differences in the Hornsea 3 figures used, and also due to the changes to the Vanguard alone predicted CRM figures due to the revised layout scenarios and increased draught heights considered by Vanguard since our Hornsea 3 Deadline 7 submission.

b. <u>Cumulative collision risk</u>

2.2.4. The Applicant has updated the EIA cumulative collision risk assessments since our Deadline 7 response in REP7-075. The figures have been updated following revised collision risk modelling (CRM) for Vanguard alone by the Applicant to account for increased draft height (as presented in AS-049). Therefore, Natural England has evaluated the cumulative CRM assessments presented by the Applicant in the updated cumulative and in-combination collision risk assessments in the Deadline 7.5 document, AS-048 for each of the five key species considered to be at risk of cumulative collision impacts: gannet, kittiwake, lesser black-backed gull (LBBG), herring gull and great black-backed gull (GBBG).

2.2.5. Table 3 shows the cumulative CRM total predictions for each of the five key species at risk of CRM for both excluding and including Hornsea 3 in the cumulative totals, as calculated by the Applicant in AS-048, which Natural England are in agreement with.

Table 3 Percentage of baseline mortality for cumulative CRM for EIA for both excluding Hornsea 3 (H3) and including Hornsea 3, using average across all age class mortality rates, as used by the Applicant

	Cumulative CRM prediction*		**Largest BDMPS (North Sea) individuals, Furness (2015)**	**% baseline mortality largest BDMPS**		**Biogeographic population individuals (Furness 2015)**	**% baseline mortality biogeographic**	
	Excl. H3	**Incl. H3****		**Excl. H3**	**Incl. H3**		**Excl. H3**	**Incl. H3**
Gannet	2,686	2,735	456,298	3.08	3.14	1,180,000	1.19	1.21
Kittiwake	3,817	4,144	839,456**	2.91	3.16	5,100,000	0.48	0.52
LBBG	513	530	209,007	1.74	1.80	864,000	0.42	0.44
Herring gull	761	770	466,511	0.94	0.95	1,098,000	0.40	0.40
GBBG	937	1,003	91,399	5.54	5.93	235,000	2.16	2.31

* Based on the Applicant's cumulative figures presented in AS-048.

** Figures included for Hornsea 3 are those used by Natural England in our Deadline 7 response during the examination for that project (Natural England 2019)

*** Population estimate for all UK colonies within North Sea BDMPS scale (from Furness 2015)

2.3. GANNET CUMULATIVE: collision risk + displacement combined

2.3.1. Natural England's calculated cumulative collision total for gannet of 2,686 birds excluding Hornsea 3 and 2,735 including Hornsea 3 exceeds 1% of baseline mortality of the North Sea BDMPS scale and the biogeographic population (Furness 2015) – the figure excluding Hornsea 3 equates to 3.08% of baseline mortality of the BDMPS and 1.19% of baseline mortality of the biogeographic population, and the figure including Hornsea 3 equates to 3.14% of the BDMPS and 1.21% of the biogeographic population baseline mortality (Table 3). This is not insignificant and requires further consideration.

2.3.2. The Applicant has considered in their assessment outputs from the Population Viability Analysis (PVA) model for the British gannet population undertaken by WWT (2012). This PVA was run over 25 years and therefore does not cover impacts from the last 5 years of the 30 year lifespan of Norfolk Vanguard (or the last 10 years of the 35 year lifespan of the Hornsea 3 project). Additionally it has not been run using the 'matched runs/pairs' approach advised by Natural England and the counterfactual metrics of population size and growth rate (as recommended by Natural England) are not presented (these issues were all highlighted in our Relevant Representations RR-106). Therefore, ideally this PVA should have been updated by the Applicant to address these issues. We also note that this PVA was undertaken using the estimated gannet population in 2004 (the most recent census available at that time), and the British gannet population has increased considerably since this time.

2.3.3. It should be noted that the figures above are for predicted collision mortalities only. The Applicant has not considered the combined impact of cumulative collision risk and cumulative displacement to gannet at the EIA scale in AS-048, and has only considered the combined in-combination impact of in-combination displacement and in-combination displacement for HRA. However, the cumulative displacement assessment has not been updated since the Applicant's Deadline 6 submission (Table 7 of REP6-021), so we note that adding predicted cumulative gannet displacement mortality would add 253-337 birds per annum including Hornsea 3 to the cumulative collision total. This gives a combined total cumulative collision plus displacement impact of up to **3,072** gannet mortalities including Hornsea 3 at the EIA scale, which equates to 3.52% of baseline mortality of the BDMPS and 1.36% of baseline mortality of the biogeographic population, which is not insignificant and requires further consideration.

2.3.4. Vanguard contributes 91 birds (66 from collision and up to 25 from displacement – see Chapter 13 of application documents, APP-337), 2.96% to this cumulative collision plus displacement total.

2.3.5. The northern gannet is classified as 'Least Concern' with respect to the potential for global extinction (BirdLife International 2018). However, at the UK scale the species is Amber listed in Birds of Conservation Concern (BoCC) 4 (Eaton et al. 2015). The BoCC Amber listing is due to:

- Localisation of breeding population within Important Bird Areas (IBAs)/Special Protection Areas (SPAs) (Eaton et al. 2015).
- International importance of UK population – threshold of 20% of global population (Eaton et al. 2015). It has been estimated that the UK holds 55.6% of the global population (JNCC 2016).

2.3.6. Based on the above conservation assessment, and given the UK's particular responsibility for gannet because of supporting over half of the global population, the predicted impacts at the North Sea population scale have the potential to give rise to significant effects and therefore **we are unable to rule out a significant (moderate adverse) effect on gannet from cumulative collision and displacement mortality at an EIA scale**.

2.4. KITTIWAKE CUMULATIVE: collision risk

2.4.1. Natural England's calculated cumulative collision total for kittiwake of 3,817 birds excluding Hornsea 3 and 4,144 including Hornsea 3 exceeds 1% of baseline mortality of all UK kittiwake colonies within the North Sea BDMPS scale (Furness 2015) – the figure excluding Hornsea 3 equates to 2.91% of baseline mortality and the figure including Hornsea 3 equates to 3.16% (Table 3). This is not insignificant and requires further consideration.

2.4.2. We note that the Applicant has used the kittiwake PVA constructed during the East Anglia 3 offshore wind farm examination for assessing the cumulative CRM impacts on the UK North Sea and Channel BDMPS population, available from Appendix 1 of EATL (2015). This PVA was run over 25 years and therefore does not cover impacts from the last 5 years of the 30-year lifespan of Norfolk Vanguard (or the last 10 years of the 35-year lifespan of the Hornsea 3 project). Additionally, it has not been run using the 'matched runs/pairs' approach advised by Natural England and it appears that only the counterfactual of population size metric is available and that the counterfactual of growth rate metric is not presented (these issues were all highlighted in our Relevant Representations RR-106). Therefore, ideally this PVA should have been updated by the Applicant to address these issues.

2.4.3. Vanguard contributes 115 collisions to the cumulative totals, this equates to 3.03% of the total of 3,817 excluding Hornsea 3 and 2.78% of the total of 4,144 including Hornsea 3.

2.4.4. Kittiwake are listed as 'Vulnerable' to global extinction on the IUCN Red List (raised from Least Concern to Vulnerable in 2017) as a result of breeding population declines in Europe of >40% over 39 years (Birdlife International 2018). Kittiwake is also listed as Red on BoCC4 (Eaton et al. 2015) as a result of severe population declines in the UK.

2.4.5. Therefore, based on the above the predicted impacts at the North Sea population scale have the potential to give rise to significant effects and therefore **we are unable to rule**

out a significant (moderate adverse) effect on kittiwake from cumulative collision mortality at an EIA scale.

2.5. LESSER BLACK-BACKED GULL (LBBG) CUMULATIVE: collision risk

2.5.1. Natural England's calculated cumulative collision total for LBBG of 513 birds excluding Hornsea 3 and 530 including Hornsea 3 exceeds 1% of baseline mortality of the North Sea BDMPS scale (Furness 2015) (which are the same as those calculated by the Applicant) – the figure excluding Hornsea 3 equates to 1.74% of baseline mortality and the figure including Hornsea 3 equates to 1.80% (Table 3). This is not insignificant and requires further consideration.

2.5.2. Vanguard contributes 23 collisions to the cumulative totals, this equates to 4.5% of the total of 513 excluding Hornsea 3 and 4.3% of the total of 530 including Hornsea 3.

2.5.3. The LBBG is classified as 'Least Concern' (BirdLife International 2018). The overall population trend across its range is increasing, although it has experienced recent declines at a UK level (Balmer et al. 2013) and the species is Amber listed in BoCC 4 (Eaton et al. 2015). Quite a high proportion of birds in the largest BDMPS of 209,007 will be UK breeding birds (Furness 2015). However, there is uncertainty in the predicted collision figures due the uncertainty/variability in the input parameters and some degree of precaution in the cumulative total regarding the nocturnal activity rate and build out scenarios. It is also worth noting that there is limited evidence and therefore some uncertainty around baseline mortality rates. **Therefore, we agree with the Applicant's conclusion in paragraph 148 of AS-048 of minor adverse impact from cumulative collision to LBBG at an EIA scale.**

2.6. HERRING GULL CUMULATIVE: collision risk

2.6.1. Natural England's calculated cumulative collision total for herring gull of 761 birds excluding Hornsea 3 and 770 including Hornsea 3 equates to 0.94% (excluding Hornsea Three) and 0.95% (including Hornsea 3) of baseline mortality of the largest BDMPS and to 0.40% (excluding and including Hornsea 3) of baseline mortality of the biogeographic population (Table 3).

2.6.2. Vanguard contributes 14 collisions to the cumulative totals, this equates to 1.84% of the total of 761 excluding Hornsea 3 and 1.82% of the total of 770 including Hornsea 3.

2.6.3. Therefore, based on the cumulative CRM figures presented in AS-048, our conclusion remains that set out in our Deadline 7 response (REP7-075) that we could conclude no significant cumulative CRM impact at the EIA scale for herring gull. **We therefore agree with the Applicant's conclusion in paragraph 109 of AS-048 of minor adverse impact from cumulative collision to herring gull at an EIA scale.** We again note that the cumulative total is now approaching 1% of baseline mortality of the largest BDMPS, reinforcing the need for herring gull CRM to have been carried out, and the need for all future offshore wind farm projects in the North Sea to do similarly.

2.7. GREAT BLACK-BACKED GULL (GBBG) CUMULATIVE: collision risk

2.7.1. Natural England's calculated cumulative collision total for GBBG of 937 birds excluding Hornsea 3 and 1,003 including Hornsea 3 exceeds 1% of baseline mortality of the North Sea BDMPS scale and the biogeographic population (Furness 2015) (which is the same as the Applicant has calculated) – the figure excluding Hornsea 3 equates to 5.54% of baseline mortality of the BDMPS and 2.16% of baseline mortality of the biogeographic population, and the figure including Hornsea 3 equates to 5.93% of the BDMPS and 2.31%

of the biogeographic population baseline mortality (Table 3). This is not insignificant and requires further consideration.

2.7.2. Vanguard contributes 47 collisions to the cumulative totals, this equates to 5.0% of the total of 937 excluding Hornsea 3 and 4.67% of the total of 1,003 including Hornsea 3.

2.7.3. GBBG is classed as ‘Least Concern’ of global extinction by IUCN. The overall population trend across its range is stable, although at a UK level the species is Amber listed in BoCC 4 (Eaton et al. 2015) due to moderate declines in both the breeding and non-breeding populations.

2.7.4. We note that the Applicant has used the GBBG PVA constructed during the East Anglia 3 offshore wind farm examination for assessing the cumulative CRM impacts on the UK North Sea and Channel BDMPS population, available from Appendix 1 of EATL (2016). This PVA was run over 25 years and therefore does not cover impacts from the last 5 years of the 30 year lifespan of Norfolk Vanguard (or the last 10 years of the 35 year lifespan of the Hornsea 3 project). Additionally it has not been run using the ‘matched runs/pairs’ approach advised by Natural England and it appears that only the counterfactual of population size metric is available and that the counterfactual of growth rate metric is not presented (these issues were all highlighted in our Relevant Representations RR-106). Therefore, ideally this PVA should have been updated by the Applicant to address these issues. However, using the existing PVA as the current best available evidence, the outputs suggest:

- If the additional mortality from the wind farms excluding Hornsea 3 is 950 (closest PVA output to the cumulative total of 937 excluding Hornsea 3) then the UK North Sea and Channel BDMPS GBBG population after 25 years will be around 6.5-8% lower than it would have been in the absence of the additional mortality using the density dependent model or around 22% lower using the density independent model.
- If the additional mortality from the wind farms including Hornsea 3 is 1,000 (closest PVA output to the cumulative total of 1,003 including Hornsea 3) then the UK North Sea and Channel BDMPS GBBG population after 25 years will be around 6.8-8.9% lower than it would have been in the absence of the additional mortality using the density dependent model or around 23% lower using the density independent model.

2.7.5. Based on the above and the cumulative CRM figures presented in AS-048, **we are unable to rule out a significant (moderate adverse) effect on GBBG from cumulative collision mortality at an EIA scale**, which is the same conclusion we reached at East Anglia 3.

2.8. LITTLE GULL CUMULATIVE: collision risk

2.8.1. We note that the Applicant has included in the cumulative/in-combination assessment Triton Knoll, Race Bank, Sheringham Shoal, Hornsea 1, Hornsea 2, Hornsea 3 and Vanguard in Table 25 of AS-048.

2.8.2. Natural England would also advise that:

- Dudgeon is also included in the in-combination assessment (note that Dudgeon has completed an assessment of collision risk for little gull at the Greater Wash SPA).
- As Vanguard is included in the in-combination assessment, we also query why the other projects in the former East Anglia zone (e.g. East Anglia 1 and East Anglia 3) are also not included.

2.8.3. We agree that the CRM figures presented for the various sites in Table 25 of AS-048 have been updated for an avoidance rate of 99.2%.

2.8.4. As noted in our general comments above on cumulative assessments, we do not consider it is appropriate to adjust the figures for the other OWFs based on build out capacities unless the reduction is legally secured and CRM re-run.

2.8.5. As we do not consider that figures have been included in the assessment from all relevant OWFs we cannot reach a conclusion regarding the significance of the level of predicted cumulative impact.

3. HABITATS REGULATIONS ASSESSMENT (HRA)

It should be noted that the general comments on the cumulative assessment in Section 1.2.1 above regarding built out layouts/capacity, nocturnal activity and the Hornsea 3 figures are also relevant to the HRA in-combination assessments.

3.1. FLAMBOROUGH & FILEY COAST (FFC) SPA: GANNET

a. <u>Vanguard alone: collision and displacement impacts</u>

3.1.1. For the impact from collision risk from Vanguard alone to gannets from the FFC SPA, we again agree with the apportionment rates used by the Applicant in AS-048 of 100% in the breeding season, 4.8% in autumn and 6.2% in spring. We also welcome that the full breeding season with adjusted migration seasons has also been presented.

3.1.2. We agree with the apportioned figure of 20 gannet collisions per annum from Vanguard alone set out by the Applicant in Table 3 of AS-048 based on the above apportionment rates and seasonal definitions for the central input values. We welcome that the Applicant has also considered the uncertainty/variability in the input data through considering in the assessment the range of collision predictions based on using the 95% confidence intervals (CIs) around the bird density data. However, as we noted previously in our Deadline 7 response (REP7-075), we again do not get the same seasonal range of figures as presented by the Applicant in paragraph 17 of AS-048: the Applicant's calculated range is 5.8-39.2 gannet collisions, whereas Natural England calculates this to be 1-56 collisions. We therefore again suggest the Applicant revisits these figures.

3.1.3. From Table 4, the predicted collision impacts presented in the Applicant's AS-048 document for the gannet feature of FFC SPA are **20 (1-56)** collisions per annum for Norfolk Vanguard alone. The predicted 20 adults per annum equates to around 1% of baseline mortality of the colony.

3.1.4. It should be noted that these figures are for predicted collision mortalities only. The Applicant has not considered the combined impact of collision risk and displacement from Vanguard alone in its submissions in AS-048, and has only considered the combined in-combination impact with other plans and projects of in-combination collision plus in-combination displacement. However, we note that adding predicted gannet displacement mortality for Vanguard alone would add 2.5-3.3 adults per annum to FFC SPA (as presented by the Applicant in Table 7 of REP6-021 (which Natural England are in agreement with) to the alone total. The Applicant has not considered the uncertainty/variability in the displacement predictions through considering the predictions using the upper and lower 95% Confidence Intervals (CIs) of the bird abundance data. Natural England calculates this range to be 0.8-1.1 birds using the lower CI data and 6.2-8.3 using the upper CI data. This gives a combined total alone impact of up to **23 (range of up to 2-64)** adult gannet mortalities from FFC from combined collision and displacement from the project alone. The predicted 23 adults per annum also equates to around 1% of baseline mortality of the colony (see Table 4).

3.1.5. Therefore, the potential impacts on the SPA require further consideration.

Table 4 Percentage of baseline mortality for impact levels for Vanguard alone for gannet for FFC SPA. Baseline mortality calculated using adult only colony size and adult mortality rate (8.8% from Horswill & Robinson 2015).

	GANNET PREDICTED MORTALITY VANGUARD ALONE, HRA: FFC SPA			
	Mortality prediction (range based on 95% CIs of density data)	% of baseline mortality of FFC SPA designated population* (used by Applicant)	% of baseline mortality of FFC SPA 2017 count** (used by Applicant)	% of baseline mortality of FFC SPA mean of 2012, 15 & 17 census data***
Based on CRM figures from Table 3 of AS-048 WCS 50% turbines in Vanguard West & 50% in Vanguard East) with increased draught height	20 (1-56)****	0.83 (0.02-2.36)	1.00 (0.03-2.86)	0.90 (0.03-2.57)
Collision + displacement for Vanguard alone	23 (2-64)*****	0.98 (0.08-2.72)	1.18 (0.1-3.29)	1.06 (0.09-2.96)

* 11,061 pairs (22,122 adults), 1% baseline mortality = 19 birds
** 13,391 pairs (26,782 adults), 1% baseline mortality = 24 birds
*** 24,594 adults, 1% baseline mortality = 22 birds. We recommend that this population size is sued in assessment of baseline mortality, as it covers the years contemporaneous with the Vanguard baseline survey data
**** Note discrepancies in figures calculated by Applicant for the range based on 95% CIs of bird density and those calculated by Natural England. The figures calculated by Natural England are presented above
***** Based on combined CRM alone figures plus displacement figures (based on WCS of 80% displacement and 1% mortality)

3.1.6. We welcome that the Applicant has considered in AS-048 the predicted collision figures for Vanguard alone with the outputs from the updated FFC SPA gannet PVA undertaken during the Hornsea 3 examination (Hornsea Project Three Offshore Wind Farm 2019). Natural England notes that we had outstanding concerns with the Hornsea 3 PVAs which were not resolved by the close of the Examination, relating to the number of simulations and the demographic data not being updated (see our Deadline 6 response to the Hornsea 3 Examination – written summary of representations of ISH5[1]). This nevertheless represents the best available evidence on which to base an assessment, though this should not be taken as an endorsement or 'acceptance' of the model.

3.1.7. There is no clear evidence to support the application of any particular form or magnitude of density dependence in the modelling, therefore Natural England has based its advice on the outputs of the density independent PVA model (as these make no assumptions about the form or strength of any density dependent effects). Therefore, Natural England has focused our conclusions on the PVA outputs from the density independent model for demographic rate set 2 (the rates Natural England considers to be the most appropriate) using a matched runs approach (as per Natural England advice) (see Table 5).

3.1.8. For the combined collision and displacement impacts to gannets from the FFC SPA from Vanguard alone, if the additional mortality from the wind farm is 25 adults per annum (closest PVA output available in Hornsea Project Three Offshore Wind Farm 2019 to

[1] Natural England (2019) Hornsea Project Three Offshore Wind Farm: Natural England Written Submission for Deadline 6 – Written Submission of Natural England's Representations at Issue Specific Hearing 5, Offshore Ecology. Available from: https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010080/EN010080-001688-Natural%20England%20-%20Written%20Submission%20of%20Natural%20England's%20Representations%20at%20Issue%20Specific%20Hearing%205%20-%20Offshore%20Ecology.pdf

Vanguard alone predicted 23 mortalities from collision and displacement combined) then the population of FFC SPA after 30 years will be 3.2% lower than it would have been in the absence of the additional mortality. The population growth rate would be reduced by 0.1% (Table 5).

3.1.9. If the upper range of collision and displacement combined of 64 birds (as calculated by Natural England) is considered, then if the additional mortality from Vanguard alone is 50-75 adults per annum (closest PVA outputs available in Hornsea Project Three Offshore Wind Farm 2019 to Vanguard upper range predicted for collision and displacement combined of 64 mortalities) then the population of FFC SPA after 30 years will be 6.4-9.4% lower than it would have been in the absence of the additional mortality and the population growth rate would be reduced by 0.2-0.3% (Table 5).

Table 5 Predicted population impacts on the gannet population of FFC SPA for the range of mortality impacts predicted for Norfolk Vanguard alone. PVA impact metrics are as provided in Hornsea Project Three Offshore Wind Farm (2019). The range of predicted project alone figures are indicated in pink. The darker shaded cells represent the level of impact closest to the central value of the prediction.

GANNET – FFC SPA VANGUARD ALONE					
Additional mortality	% Baseline Mortality using designation population size (22,122 adults), as used by Applicant	% Baseline Mortality using 2017 count size (26,782 adults), as used by Applicant	% Baseline Mortality using mean of 2012, 15 & 17 census data (24,594 adults)	Counterfactual of Final Population Size (CPS)*	Counterfactual of Growth rate (CGR)**
5	0.26	0.21	0.23		No value available
10	0.51	0.42	0.46		No value available
20	1.03	0.85	0.92		No value available
25*	1.28	1.06	1.16	0.968 (0.967-0.968)	0.999
30	1.54	1.27	1.39		No value available
40	2.05	1.70	1.85		No value available
50*	2.57	2.12	2.31	0.936 (0.936-0.937)	0.998
75	3.85	3.18	3.47	0.906 (0.905-0.907)	0.997
100	5.14	4.24	4.62	0.877 (0.876-0.878)	0.995

* Gannet, demographic rate set 2, counterfactuals of population size after 30 years, estimated using a matched runs method, from 1000 density independent simulations. See Table A2_3.1 in Hornsea Project Three Offshore Wind Farm (2019)
** Gannet, demographic rate set 2, counterfactuals of population growth rate after 35 years, estimated using a matched runs method, from 1000 density independent simulations. See Table A2_3.3 in Hornsea Project Three Offshore Wind Farm (2019). Whilst Vanguard's lifespan is 30 years, data on counterfactuals of growth rate are only available in Hornsea Project Three Offshore Wind Farm (2019) for after 35 years.

3.1.10. The gannet population of FFC SPA increased at 11.1% per annum (between 2003/4 and 2015, JNCC Seabird Monitoring Programme SMP data). Using FFC SPA census data 2002-2017 the growth rate was 9.4% per annum. However, it is not known what the growth rate of the colony will be over the next 30 years and this should therefore be considered when judging the significance of predicted impacts against the conservation objectives for the feature.

3.1.11. Natural England has reviewed growth rates for the 22 gannet colonies across Britain, Channel Islands and Ireland with repeated census data (Cramp et al. 1974, Lloyd et al. 1991, Mitchell et al. 2004, plus more recent count data from the SMP). The Flamborough/Bempton gannet colony was founded in the late 1930s (Cramp et al. 1974) and so has been in existence now for about 80 years. Thus, by the end of 30 years of Vanguard it will be about 110 years in age. Given the analysis of trends in gannet colony growth rates amongst a suite of long-established colonies, it is highly likely that its annual growth rate averaged over the whole period since founding will drop from its current average of c 11% over the first 80 years. The highest annual colony growth rate calculated over a period of >100 years is 4.5% at Grassholm. The Flamborough colony is unlikely to achieve a higher annual growth rate than this. The average annual growth rate calculated over a period of >90 years across the 8 gannet colonies with records exceeding 90 years is 1.8%. Amongst these colonies the mean annual growth rate over the most recent years of their records (80+ years) has been just 1.2% per annum (or 1.3% excluding Sula Sgeir (as the growth rate here may be influenced adversely by an annual licenced harvest of young birds)) compared to an average rate of 2.0% per annum during the first 80 or so years of their existence. Therefore, Natural England has considered the counterfactuals of final population size for the predicted levels of alone additional mortality for a range of plausible future growth rate scenarios for FFC of 1, 2, 3, 4 and 5% per annum.

3.1.12. The Conservation Objective for the gannet population of the FFC SPA is to maintain the size of the breeding population at a level which is above 8,469 pairs (16,938 adults), whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent. The latest mean count is 24,594 adults based on the mean of the 2012, 2015 and 2017 counts.

3.1.13. For the predicted Vanguard alone collision plus displacement mortality to FFC SPA gannets of 23 (range 2-64) mortalities per year, from the closest updated PVA outputs in Hornsea Project Three (2019) of 25 and 75 additional mortalities, the colony would still be predicted to grow above the current mean population of 24,594 adults under any growth rate scenario from 1% to up to 5%. This would allow the conservation objective to be met and therefore **no AEOI of the gannet feature of the FFC SPA can be concluded for collision plus displacement impacts from Vanguard alone**.

b. <u>In-combination collision risk and displacement impacts with other plans and projects</u>

3.1.14. As noted in our Deadline 7 response (REP7-075), we again welcome that for gannet for the FFC SPA all of the other offshore wind farm collision and displacement predictions for autumn and spring in Table 5 of AS-048 have been apportioned using the Natural England recommended rates of 4.8% in autumn and 6.2% in spring. We also again welcome that figures for the Hywind, Kincardine and Moray West offshore wind farms (OWFs) are again included. We also welcome that the in-combination assessment in AS-048 makes reference to the updated PVA undertaken for Hornsea 3, but note our above comments regarding the outstanding with this PVA**.**

3.1.15. The in-combination collision total calculated by Natural England is 212 gannets from the FFC SPA per annum excluding Hornsea 3 and 230 including Hornsea 3 (Natural England has used the same figures for Hornsea 3 as we used in Natural England 2019). These predicted in-combination collision impacts equate to more than 1% of baseline mortality of the colony (see Table 6).

3.1.16. Again, these figures are for predicted collision mortalities only. The Applicant has considered the predicted impacts to gannets from the FFC SPA of in-combination collision and in-combination displacement combined in Section 3.1.2.1 of AS-048. Adding predicted displacement mortality would add 49-65 birds to the in-combination total. This gives a combined total in-combination collision plus displacement impact of up to 212+65=**277 (excluding Hornsea 3)** or 230+65=**295 (including Hornsea 3)**. The predicted impacts again equates to more than 1% of baseline mortality of the colony (see Table 6).

Table 6 Percentage of baseline mortality for in-combination impact levels for excluding and including Hornsea 3 (H3) for gannet for FFC SPA. Baseline mortality calculated using adult only colony size and adult mortality rate (8.8% from Horswill & Robinson 2015).

	GANNET PREDICTED IN-COMBINATION MORTALITY, HRA: FFC SPA			
	Mortality prediction	% of baseline mortality of FFC SPA designated population* (used by Applicant)	% of baseline mortality of FFC SPA 2017 count** (used by Applicant)	% of baseline mortality of FFC SPA mean of 2012, 15 & 17 census data***
In-combination CRM, based on figures from Table 5 of AS-048	212 excl. H3 230 incl. H3****	10.87 excl. H3 11.79 incl. H3	8.98 excl. H3 9.74 incl. H3	9.78 excl. H3 10.61 incl. H3
In-combination collision + in-combination displacement*****	277 excl. H3 295 incl. H3****	11.84 excl. H3 15.15 incl. H3	9.78 excl. H3 12.52 incl. H3	10.65 excl. H3 13.63 incl. H3

* 11,061 pairs (22,122 adults), 1% baseline mortality = 19 birds
** 13,391 pairs (26,782 adults), 1% baseline mortality = 24 birds
*** 24,594 adults, 1% baseline mortality = 22 birds. We recommend that this population size is sued in assessment of baseline mortality, as it covers the years contemporaneous with the Vanguard baseline survey data
**** Figures included for Hornsea 3 collisions are those used by Natural England in our Deadline 7 response during the examination for that project (Natural England 2019)
***** In-combination displacement figure used in total is that for WCS of 80% displacement and 1% mortality

3.1.17. We welcome that the Applicant has considered in AS-048 the predicted in-combination collision figures and the combined in-combination collision plus displacement figures with the outputs from the updated FFC SPA gannet PVA undertaken during the Hornsea 3 examination (Hornsea Project Three Offshore Wind Farm 2019), though we again note our comments above regarding outstanding issues with this PVA.

3.1.18. For the reasons set out above, Natural England has again focused our conclusions on the PVA outputs from the density independent model for demographic rate set 2 using a matched runs approach (see Table 7).

3.1.19. For the combined collision and displacement impacts in-combination with other plans and projects, if the additional mortality from the offshore wind farms is 275-300 per annum (closest PVA outputs to the combined in-combination displacement and collision mortality figures of 277 excluding Hornsea 3 and 295 including Hornsea 3) then the population of FFC SPA after 30 years will be 30.4-32.7% lower than it would have been in the absence of the additional mortality. The population growth rate would be reduced by 1.2-1.4% (Table 7).

Table 7 Predicted Population impacts on the gannet population of FFC SPA for the range of mortality impacts predicted for in-combination collision and in-combination collision plus in-combination displacement. PVA Impact Metrics are as provided in Hornsea Project Three Offshore Wind Farm (2019). The range of predicted figures are indicated in purple. The darker shaded cells represent the level of impact closest to the combined in-combination collision plus in-combination displacement predictions

GANNET	FFC SPA				
Additional mortality	% Baseline Mortality using designation population size (22,122 adults), as used by Applicant	% Baseline Mortality using 2017 count size (26,782 adults), as used by Applicant	% Baseline Mortality using mean of 2012, 15 & 17 census data (24,594 adults)	Counterfactual of Final Population Size (CPS)*	Counterfactual of Growth rate (CGR)**
225	11.56	9.55	10.40	0.743 (0.741-0.746)	0.990
250	12.84	10.61	11.55	0.719 (0.716-0.722)	0.989
275	14.13	11.67	12.71	0.696 (0.693-0.698)	0.988
300	15.41	12.73	13.86	0.673 (0.670-0.676)	0.986
325	16.69	13.79	15.02	0.651 (0.648-0.654	0.985

* Gannet, demographic rate set 2, counterfactuals of population size after 30 years, estimated using a matched runs method, from 1000 density independent simulations. See Table A2_3.1 in Hornsea Project Three (2019)
** Gannet, demographic rate set 2, counterfactuals of population growth rate after 35 years, estimated using a matched runs method, from 1000 density independent simulations. See Table A2_3.3 in Hornsea Project Three (2019).

3.1.20. As noted in the assessment of impacts from Vanguard alone above (Section 2.1.1), it is not known what the growth rate of the colony will be over the next 30 years and this should be considered when judging the significance of predicted impacts against the conservation objectives for the feature.

3.1.21. Natural England has used the same review of gannet colony growth rates as used in the alone assessment and has again considered the counterfactuals of final population size for the predicted levels of in-combination additional mortality for a range of plausible future growth rate scenarios for FFC of 1, 2, 3, 4 and 5% per annum.

3.1.22. The Conservation Objective for the gannet population of the FFC SPA is to maintain the size of the breeding population at a level which is above 8,469 pairs (16,938 adults), whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent. The latest mean count is 24,594 adults based on the mean of the 2012, 2015 and 2017 counts.

3.1.23. For the predicted in-combination with other plans and projects collision plus displacement mortality to FFC SPA gannets of 277 mortalities per year excluding Hornsea 3, from the closest updated PVA output in Hornsea Project Three (2019) of 275 additional mortalities, the colony would be predicted to reduce from its current size of 24,594 adults for a growth rate of 1%, but would still be above the size of the 8,469 pairs or 16,938 adults. The colony would be predicted to continue to grow above the current mean population of 24,594 adults under any growth rate scenario from 2% to up to 5% per annum.

3.1.24. If the colony were to experience an annual growth rate of 2% or more per annum over the next 30 or so years, then the integrity of the site for this feature is high, with high rates for self-repair, and self-renewal under dynamic conditions with minimal external management. Therefore, the FFC Gannet Population is believed to be robust enough to allow the conservation objective to maintain the population at (or above) designation levels and sustain additional alone and in-combination mortalities from the offshore wind farms. Our justification for this position is we consider it to be highly unlikely that the FFC annual growth rate would be as low as 1%, and from the analysis of gannet colony growth rates we have conducted the current annual growth rate of c 11% appears to be relatively high for a colony of this age and so the colony is likely to do better than a 1.3 % annual growth rate in the foreseeable future.

c. Overarching summary for FFC Gannet in-combination.

3.1.25. Natural England advises that on the above information that an AEOI of the gannet feature of the FFC SPA can be ruled out for collisions plus displacement impacts from in-combination with other plans and projects if Hornsea 3 is excluded from the in-combination total.

3.1.26. **However, due to Natural England's significant concerns regarding the incomplete baseline surveys for the Hornsea 3 project, and the associated level of uncertainty as regards the potential impacts of that project, Natural England is not in a position to advise that an AEOI can be ruled out for the gannet feature of the FFC SPA for collision plus displacement impacts in-combination with other plans and projects when Hornsea 3 is included in the in-combination total.**

3.2. FLAMBOROUGH & FILEY COAST (FFC) SPA: KITTIWAKE

a. Vanguard alone: collision impacts

3.2.1. We note that in AS-048 the HRA for kittiwake at the FFC SPA for Vanguard alone appears to be based on collision predictions for the breeding season at the Vanguard West site combined with the collision predictions across the worst case scenario layout option b (50% turbines in Vanguard West and 50% in Vanguard East) across both sites in the migration seasons. This is because the Applicant considers the Vanguard West site is closer to the FFC SPA and there is more compelling evidence for connectivity on this site. It would assist the Examining Authority if the Applicant could clarify this point.

3.2.2. If this is the case Natural England has significant concerns regarding this approach, as this will not be the realistic worst case scenario for the Vanguard Rochdale envelope, as using the full breeding season and the Applicant's apportionment rates of 26.1% in the breeding season, 5.4% in autumn and 7.2% in spring to apportion collisions from the worst case scenario layout option of 50% of turbines in Vanguard West and 50% in Vanguard East results in a higher annual collision prediction of kittiwakes from the FFC SPA than the Applicant's approach in AS-048. Given that there is evidence from the tracking data for connectivity of kittiwakes from the FFC SPA with the Vanguard sites during the breeding season, we consider the full breeding season with adjusted migration seasons to be the appropriate seasonal definitions to use for this assessment.

3.2.3. The Applicant has continued to consider that the calculated apportionment of 26.1% for the breeding season is precautionary and has not considered the approach of a matrix as advised by Natural England at the telecall with the Applicant on 2nd April 2019 and as advised in our Deadline 7 response (REP7-075).

3.2.4. Based on the above, our advice regarding collision impacts to kittiwake from the FFC SPA from Vanguard alone remains that presented in our Deadline 7 response, REP7-075. However, our calculations have been updated to reflect the revised collision figures from Vanguard alone based on the increased draft height as presented in AS-048.

3.2.5. Natural England has considered the apportionment of kittiwake collisions to the FFC SPA from Vanguard alone using what is likely to be a precautionary 86% apportioning rate in the breeding season together with the agreed 5.4% in autumn and 7.2% in spring. This assessment has been made by applying these apportionment rates to the CRM predictions for the revised worst case layout of 50% of the turbines in Vanguard West and 50% in East together with the increased draft height (as set out in AS-049). Using these rates results in annual total of **43 kittiwake collisions (range of 2-120 based on 95% CIs of density data) to the FFC SPA**. These figures equate to 0.33% (range 0.02-0.93%) of baseline mortality of the FFC SPA kittiwake colony using the designated colony adult population or to 0.29% (range 0.02-0.80%) of baseline mortality using the mean of 2016-17 population and an adult mortality rate of 14.6% (Horswill & Robinson 2015). It is worth noting that there is limited evidence and therefore some uncertainty around baseline mortality rates. On the basis of these figures, **Natural England advises that a conclusion of no AEOI of the**

kittiwake feature of the FFC SPA from collision risk from Norfolk Vanguard alone can be reached.

b. **In-combination collision risk impacts with other plans and projects**

3.2.6. As noted in our Deadline 7 response (REP7-074), we again welcome that for kittiwake for the FFC SPA all of the other offshore wind farm collision predictions for autumn and spring in Table 12 of AS-048 have been apportioned using the Natural England recommended rates of 5.4% in autumn and 7.2% in spring and that the breeding season apportionment rates labelled as the 'NE method'[2] from the East Anglia 3 assessment have been used with the higher rate of 83% also used for Hornsea 2. We also again welcome that figures for the Hywind, Kincardine and Moray West offshore wind farms (OWFs) are again included. We welcome that the in-combination assessment in AS-048 makes reference to the updated PVA undertaken for Hornsea 3, though as with the gannet PVA, Natural England notes that we had outstanding concerns with the Hornsea 3 PVAs which were not resolved by the close of the Examination, relating to the number of simulations and the demographic data not being updated (see Deadline 6 response to the Hornsea 3 Examination – written summary of representations of ISH5). This nevertheless represents the best available evidence on which to base an assessment, though this should not be taken as an endorsement or 'acceptance' of the model.

3.2.7. **The in-combination collision total calculated by the Applicant is 332 kittiwakes from the FFC SPA per annum excluding Hornsea 3 and 490 including Hornsea 3**. Natural England calculates the annual apportioned figure to the FFC SPA from Vanguard alone to be higher than the Applicant at 43 collisions per year for using a precautionary 86% breeding season apportionment rate and the Applicant's rates of 5.4% in autumn and 7.2% in spring all on the figures for the worst case scenario layout of 50% of turbines in Vanguard West and 50% in Vanguard East. We have also used the same annual figure of 181 kittiwakes apportioned to the FFC SPA for Hornsea 3 as used in our Deadline 7 response during the Hornsea 3 examination (Natural England 2019) **Using these figures in the in-combination assessments brings the total figures to 366 kittiwake collisions excluding Hornsea 3 and 547 including Hornsea 3**. Both the Applicant's calculated in-combination figures and those calculated by Natural England equate to more than 1% of baseline mortality of the colony (see Table 8).

Table 8 Percentage of baseline mortality for in-combination collision impacts for excluding and including Hornsea 3 (H3) for kittiwake for FFC SPA. Baseline mortality calculated using adult only colony size and adult mortality rate (14.6% from Horswill & Robinson 2015).

	KITTIWAKE PREDICTED IN-COMBINATION CRM MORTALITY, HRA: FFC SPA		
	Mortality prediction	% of baseline mortality of FFC SPA designated population* (used by Applicant)	% of baseline mortality of FFC SPA mean 2016-17 census data**
Applicant's in-combination CRM, based on figures from Table 12 of AS-048	332 excl. H3 490 incl. H3	2.55 excl. H3 3.77 incl. H3	2.22 excl. H3 3.27 incl. H3
Natural England's calculated in-combination CRM, based on preferred apportionment rates and CRM figures for Vanguard & same H3 figures as used in Natural England (2019)	366 excl. H3 547 incl. H3	2.81 excl. H3 4.20 incl. H3	2.44 excl. H3 3.65 incl. H3

* 89,040 adults, 1% baseline mortality = 130 birds
** 102,536 adults, 1% baseline mortality = 150 birds

[2] It should be noted that this is not an "NE method" but an interpretation of HOW2's figures using parameters that were more closely aligned with our advice than the ones the Applicant had used.

3.2.8. There is no clear evidence to support application of any particular form or magnitude of density dependence in the modelling, therefore Natural England has based our advice on the outputs of the DI models (as these make no assumptions about the form of strength of any density dependent effects). Therefore, Natural England has focused our conclusions on the PVA outputs from the density independent model for demographic rate set 2 using a matched runs approach (see Table 9).

Table 9 Predicted population impacts on the kittiwake population of FFC SPA for the range of mortality impacts predicted for Norfolk Vanguard in-combination with other plans and projects. PVA impact metrics are as provided in Hornsea Project Three Offshore Wind Farm (2019). The range of predicted in-combination figures are indicated in purple. The darker shaded cells represent the level of impact closest to the in-combination predictions in Table 8.

KITTIWAKE	FFC SPA			
Additional mortality	% Baseline Mortality using designation population size (89,040 adults)	% Baseline Mortality using mean 2016-17 census data (102,536 adults)	Counterfactual of Final Population Size (CPS)*	Counterfactual of Growth rate (CGR)**
300	2.31	2.00	0.907 (0.906-0.908)	0.997
350	2.69	2.34	0.892 (0.891-0.893)	0.996
400	3.08	2.67	0.878 (0.877-0.879)	0.996
450	3.46	3.01	0.863 (0.862-0.865)	0.995
500	3.85	3.34	0.849 (0.848-0.851)	0.994
550	4.23	3.67	0.835 (0.834-0.837)	0.994

* Kittiwake, demographic rate set 2, counterfactuals of population size after 30 years, estimated using a matched runs method, from 1000 density independent simulations. See Table A2_7.1 in Hornsea Project Three Offshore Wind Farm (2019)
** Kittiwake, demographic rate set 2, counterfactuals of population growth rate after 35 years, estimated using a matched runs method, from 1000 density independent simulations. See Table A2_7.3 in Hornsea Project Three Offshore Wind Farm (2019). Whilst Vanguard's lifespan is 30 years, data on counterfactuals of growth rate are only available in Hornsea Project Three Offshore Wind Farm (2019) for after 35 years. No CLs given as they are the same as the median values.

3.2.9. If the additional mortality from the windfarm is 350 adults per annum (closest PVA outputs available in Hornsea Project Three Offshore Wind Farm 2019 to Applicant's predicted 332 mortalities for in-combination total excluding Hornsea 3 and to the 366 in-combination total calculated by Natural England using our preferred approach for Vanguard alone figures) then the population of FFC SPA after 30 years will be 10.8% lower than it would have been in the absence of the additional mortality. The population growth rate would be reduced by 0.4% (Table 9). If it is assumed that the population is stable then this would mean that the population would be 10.8% lower than the current population size. This would be counter to the restore conservation objective for this feature at the site and would result in an adverse effect on the integrity of the site. Vanguard's contribution to the in-combination total excluding Hornsea 3 based on the Applicant's figures is 2.89%, whilst Vanguard's contribution to the in-combination total excluding Hornsea 3 based on Natural England's figures is 11.76%.

3.2.10. If the additional mortality from the windfarm is 500-550 adults per annum (closest PVA outputs available in Hornsea Project Three Offshore Wind Farm 2019 to Applicant's predicted 490 mortalities for in-combination total including Hornsea 3 and to the 547 in-combination total calculated by Natural England using our preferred approach for Vanguard alone figures and the Hornsea 3 figures used in Natural England 2019) then the population of FFC SPA after 30 years will be 15.1-16.5% lower than it would have been in the absence of the additional mortality. The population growth rate would be reduced by 0.6% (Table 9). If it is assumed that the population is stable then this would mean that the population would be 15.1-16.5% lower than the current population size. This would be counter to the restore conservation objective for this feature at the site. Vanguard's contribution to the in-combination total including Hornsea 3 based on the Applicant's figures is 1.96%, whilst Vanguard's contribution to the in-combination total including Hornsea 3 based on Natural England's figures is 7.86%.

3.2.11. It is not known what the growth rate of the colony will be over the next 30 years and this should be considered when judging the significance of predicted impacts against the conservation objectives for the feature. There has been a 2.2% per annum decline in numbers for Flamborough Head and Bempton Cliffs colony[3] between 1987 and 2017 (a growth rate of 0.979 per annum). Over the period 2000 to 2017 the population has shown a 0.37% per annum increase in numbers (a growth rate of 1.0037 per annum) based on census counts in SMP (JNCC 2016).

3.2.12. Across colonies in the UK the kittiwake population declined by 44% between 1998/2000 and 2015. Between the SCR Census (1985–88) and Seabird 2000 (1998–2002) for major colonies in Britain, no sites showed a per annum increase that exceeded 4.5% (see Section B of Natural England's Deadline 4 submission for Hornsea Project 2[4]). The growth rate of the colony at Bempton/Flamborough between 2000 and 2017 was 0.37% per annum, following declines from 1987. So, it seems reasonable to assume that the FFC SPA colony growth rate is <1% per annum. Therefore Natural England has considered the counterfactuals of final population size for the predicted levels of in-combination additional mortality for a range of plausible future growth rate scenarios for FFC of stable, 0.37, 1, and 3% per annum.

3.2.13. The Conservation Objective for the kittiwake population of the FFC SPA is to restore the size of the breeding population at a level which is above 83,700 breeding pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent.

3.2.14. If we assume a 1% per annum growth rate then 350 additional mortalities per annum would result in the population being approximately 15,000-16,000 birds lower than without the additional mortality after 30 years and it would take over an additional 30 years to reach the target population compared to the no windfarm mortality scenario. If we assume a 1% per annum growth rate then 500-550 additional mortalities per annum would result in the population being around 20,000-25,000 birds lower than without the additional mortality after 30 years and it would take over an additional 70 years to reach the target population compared to the no windfarm mortality scenario. It is not possible to rule out AEOI for these scenarios.

3.2.15. If the kittiwake population were to grow at the a rate of 3% per annum over the next 30 years, then additional mortality of 500-550 per annum would result in the population being approximately 40,000 birds lower than without the additional mortality after 30 years and it would take over an additional 4 years to reach the target population compared to the no windfarm mortality scenario. In the context of a population trajectory that is currently stable or increasing at <1% per annum an additional mortality of 500-550 adults per annum over 30 years causing a reduction in growth rate of 0.4% would further harm the population and

[3] It should be noted that the new Flamborough and Filey Coast SPA includes additional cliff areas at Filey which support kittiwake but were not previously monitored as part of the SPA, hence the reference to Flamborough Head and Bempton Cliffs.

[4] Natural England (2015) Hornsea Project Two Offshore Wind Farm – Written Submission for Deadline 4. Available from: https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010053/EN010053-001163-Natural%20England.pdf

make it more difficult to restore the population to a favourable condition. Natural England is therefore currently unable to advise beyond reasonable scientific doubt that this level of impact would not be an AEOI.

3.2.16. There is no evidence to suggest that the future population trend will be significantly different from the current trend of 0.37% per annum (2000-2017), for example productivity at the colony has not been increasing in recent years (see Figure 1) (Aitken et al. 2017). So, based on the review of growth rates above, it seems reasonable to assume that the FFC SPA colony growth rate will be <1% per annum.

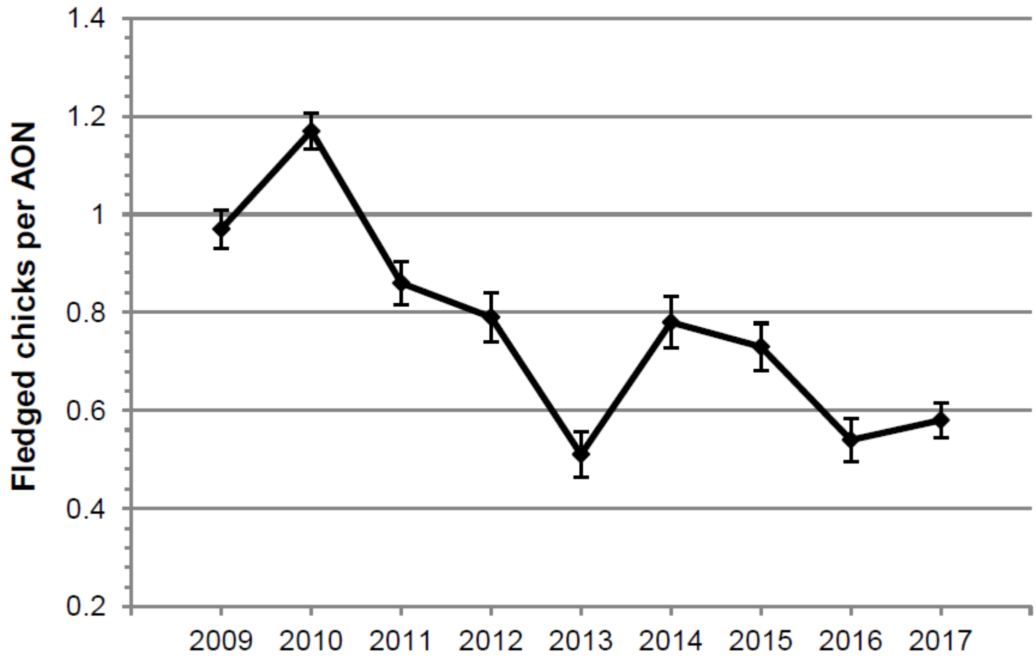

Figure 1 Flamborough/Bempton Black-legged kittiwake productivity 2009-2017, mean of plot results +/- SE. From Aitken et al. (2017). Note this does not include productivity data for Filey, where productivity is lower (e.g. in 2017 mean productivity for kittiwake at Filey was 0.39 (SE ± 0.0742) chicks per AON).

3.2.17. Therefore, as this feature has a restore conservation objective, and because there are indications that the predicted level of mortality would mean the population could decline from current levels should it currently be stable, it is not possible to rule out AEOI of the kittiwake feature of the FFC SPA for collision impacts from in-combination with other plans and projects, both including and excluding Hornsea 3.

3.3. ALDE-ORE ESTUARY SPA: LESSER BLACK-BACKED GULL

a. <u>Vanguard alone: collision impacts</u>

3.3.1. The Applicant has continued to consider a breeding season apportionment rate for LBBG to the Alde-Ore Estuary SPA of 17% and has again not considered the approach of a matrix as advised by Natural England at the telecall with the Applicant on 2nd April and as advised in our Deadline 7 response (REP7-075) or focused on the range 10-30% as advised in our Deadline 7 response (REP7-075).

3.3.2. As noted in our Deadline 7 response (REP7-075), all of the information provided by the Applicant indicates just how variable the ecology of this species can be, both between individuals within a colony and between seasons and years. As a result, it is difficult to have much confidence in pinning down an actual figure for use in apportionment. Therefore, we have based our calculations of impact from Vanguard alone in Table 10 on use of a range of breeding season apportionment rates of 10-30%, including the Applicant's preferred rate of 17% and using the revised CRM figures from AS-048. Whilst the Applicant's calculated apportionment rates for the non-breeding seasons of 3.3% in autumn and spring and 5% in winter have not been calculated from Natural England's standard approach (as highlighted in our Relevant Representations, RR-106), the Applicant's approach does not appear to make a significant difference to the apportionment figures that result from taking the Natural England recommended approach and therefore, we are content with the rates used by the Applicant for the non-breeding seasons and have used these in the calculations in Table 10.

Table 10 Percentage of baseline mortality for impact levels for LBBG for the Alde-Ore Estuary SPA, using a range of breeding season apportionment rates from 10-30% advised by Natural England and the Applicant's apportionment rates in the non-breeding seasons of 3.3% in autumn and spring and 5% in winter. Baseline mortality calculated using adult colony size and adult mortality rate (11.5% from Horswill & Robinson 2015). Grey shaded cells represent scenarios equating to more than 1% baseline mortality

		Impact collisions per annum to Alde-Ore SPA	% of baseline mortality of Alde-Ore SPA population of 2,000 pairs as used by Applicant **
Based on CRM figs in Table 16 of AS-048, using 10% breeding season apportionment*	Lwr 95% CI density	0.1	0.02
	Central	2	0.40
	Upr 95% CI density	5	1.11
Based on CRM figs in Table 16 of AS-048, using 17% breeding season apportionment (as used by Applicant)*	Lwr 95% CI density	0.2	0.04
	Central	3	0.64
	Upr 95% CI density	8	1.74
Based on CRM figs in Table 16 of AS-048, using 30% breeding season apportionment*	Lwr 95% CI density	0.3	0.07
	Central	5	1.08
	Upr 95% CI density	13	2.90

* Note that Natural England does not agree with the Applicant's figures for the 95% CIs at EIA and therefore the range of CRM predictions are based on apportionment on the Natural England calculated range of figures using the 95% CIs of density for the EIA CRM assessment
** 2,000 pairs (2007-2014), 4,000 adults. 1% baseline mortality = 4.6 birds

3.3.3. Based on the above, considering the apportionment of LBBG collisions to the Alde-Ore Estuary SPA from Vanguard alone using a precautionary upper apportioning rate in the breeding season of 30% together with the Applicant's rates of 3.3% in autumn and spring and 5% in winter, results in annual total of **5 LBBG collisions (range of 0.3-13 based on 95% CIs of density data) to the Alde-Ore Estuary SPA**. These figures equate to 1.08% (range 0.07-2.90%) of baseline mortality of the Alde-Ore Estuary SPA LBBG colony using a colony population of approximately 2,000 pairs (2007-2014) as used by the Applicant and an adult mortality rate of 11.5% (Horswill & Robinson 2015). Therefore, the potential impacts on the SPA require further consideration.

3.3.4. If the Applicant's rate of 17% apportionment in the breeding season is used with the non-breeding season rates, the predicted impacts are a total of **3 LBBG collisions (range of 0.2-8 based on 95% CIs of density data) to the Alde-Ore Estuary SPA**. These figures equate to 0.64% (range 0.04-1.74%) of baseline mortality of the Alde-Ore Estuary SPA LBBG colony. Whilst the central value equates to less than 1% of baseline mortality, the collision predictions based on the upper 95% CI of the density data does equate to more than 1% of baseline mortality of the Alde-Ore Estuary SPA colony.

3.3.5. Natural England welcomes that the Applicant has updated in REP7-063 the LBBG Alde-Ore Estuary PVA to address Natural England's comments on the previous version submitted at Deadline 6 (REP6-020). With regard to the updated version of this PVA in REP7-063, we note the following:

i. We welcome that the updated version of the PVA in REP7-063 has now been run over 5,000 simulations as recommended by Natural England.

ii. It appears that the Applicant has the column headings for the median and lower CIs the wrong way around in the tables for the counterfactuals of growth rates (CGRs) in Tables 3 and 5 of REP7-063. This has been confirmed by the Applicant in an email to Natural England dated 20th May 2019.

iii. It appears there is an error in the figure heading for Figure 3 and that this should be the counterfactuals of population size for the density dependent simulations rather than the density independent simulations. This has been confirmed by the Applicant in an email to Natural England dated 20th May 2019.

iv. With regard to the adjustment of national mean LBBG productivity with a figure to take account of the proportion of birds that miss breeding each year (in an average LBBG population), as we have no information on the proportion of birds that don't breed in any year for the Alde-Ore colony rather than try to adjust a national productivity figure to "account" for a crude estimate for the proportion of birds that don't breed in a particular year it would be better to actually use productivity rates that have been measured for the colony – i.e. to use the colony specific rates, as this seems the most evidence based approach. There will then need to be a caveat that not all birds may breed in a given year so actual productivity across the population may be lower. It looks as though there is colony specific information on productivity available and we would therefore advise using this rather than an adjusted national level.

v. We previously noted that in the version of the PVA submitted at Deadline 6 (REP6-020) we could not check the 10% baseline growth figure from the outputs presented. Whilst the Applicant now say in the Deadline 7 version (REP7-063) that this growth figure was incorrect and baseline growth rate for the density independent model is -2%, we again cannot check this from the outputs presented as the version in REP7-063 contains the same graphs and tables of counterfactuals as presented in REP6-020.

vi. We note that in the version of the PVA submitted at Deadline 6 (REP6-020) the Applicant was saying the models predicted a 10% per annum growth rate under baseline conditions and now in the updated version submitted at Deadline 7 (REP7-063) they are saying it was actually a 2% per annum decline, but state that this doesn't affect the counterfactuals. This is perhaps not unexpected, given that one merit of Natural England's preferred counterfactuals is that they are relatively robust to mis-specification of e.g. growth rates. This confusion emphasises the relevance of our request at Deadline 7 (REP7-075) for the Applicant to provide clear information about the model input and output parameters such as what the baseline growth rate in the models is, so its performance can be evaluated in detail. Furthermore, this clarification does undermine some of the assertions made by the Applicant in their Deadline 6 document (REP6-020), which stated: *"Although the trend in the Alde-Ore Estuary population is not well known, and allowing for the potential limitations in the data as noted above, the demographic rates indicate that under baseline conditions the population growth rate would be in excess of 10%. While this estimate must be treated with caution, it does indicate that smaller reductions in the growth rate, such as up to 3% for example, are unlikely to trigger a population decline. Thus, using the more precautionary density independent model, the results suggest that an adult mortality of up to 120, which corresponds to a 3% reduction in growth rate, is unlikely to trigger a population decline."*

i. Natural England would take an alternative view, namely that a reduction in growth rate of 3% has the potential to be significant for a population that already has a negative growth rate of a 2% per annum decline. Natural England also notes that there is no evidence that the LBBG Alde-Ore Estuary SPA population could or is likely to experience a 10% per annum growth rate over the next 30 years.

3.3.6. However, whilst we still have some outstanding concerns/queries regarding the updated PVA, given the examination timescales we have nevertheless considered the predicted collision figures for Vanguard alone with the outputs from the updated Alde-Ore Estuary SPA LBBG PVA in REP7-063 (see Table 11 below), as this represents the best model currently available for the assessment. Please note that this should in no way be interpreted as Natural England endorsing or 'accepting' all elements of the PVA.

3.3.7. Given that there is no evidence of density dependence operating on the LBBG Alde-Ore Estuary colony or of how it is operating, Natural England has focused our conclusions on the PVA outputs from the density independent model.

Table 11 Predicted population impacts on the LBBG population of the Alde-Ore Estuary SPA for the range of mortality impacts predicted for Norfolk Vanguard alone using 10-30% apportionment in the breeding season and agreed rates of 3.3% in autumn and spring and 5% in winter. PVA impact metrics are as provided in the Applicant's Deadline 7 updated LBBG Alde-Ore Estuary SPA PVA (REP7-063). The range of predicted project alone figures are indicated in pink. The darker shaded cells represent the level of impact closest to the central values of the prediction for the range of apportionment scenarios considered above

LBBG – ALDE-ORE ESTUARY SPA VANGUARD ALONE			
Additional mortality	% Baseline Mortality using population size of 4,000 adults (2007-2014), as used by the Applicant	Density Independent Model	
		Counterfactual of Final Population Size (CPS) after 30yrs – see Table 2 of REP7-063	Counterfactual of Growth rate (CGR) after 30yrs – see Table 3 of REP7-063*
5	1.09	0.966 (0.893-1.046)	0.999 (0.996-1.002)
10	2.17	0.930 (0.858-1.006)	0.997 (0.994-1.000)
15	3.26	0.897 (0.828-0.969)	0.996 (0.993-0.999)

* The Applicant has confirmed that the headings for the median and lower CIs are the wrong way around in REP7-063. So, we have presented the figures the correct way around above

3.3.8. If the additional mortality from Vanguard alone is 5 adults per annum (closest PVA outputs available in REP7-063 to Applicant's apportionment approach of 3 predicted adult mortalities and to the Natural England precautionary apportionment approach of 5 predicted adult mortalities, based on the mean density CRM predictions) then the population of the Alde-Ore Estuary SPA after 30 years will be 3.4% lower than it would have been in the absence of the additional mortality using the density independent model outputs and 1.1% lower using the density dependent model outputs. The population growth rate would be reduced by 0.1% using the density independent model and would not be reduced using the density dependent model (Table 11).

3.3.9. Taking account of uncertainty/variability in the CRM input parameters (using the upper 95% CI of the bird density data, as this accounts for the greatest variability in the predictions), if the additional mortality is 10-15 adults per annum (closest PVA outputs available in REP7-063 to Applicant's apportionment approach of 8 predicted adult mortalities and to the Natural England precautionary apportionment approach of 13 predicted adult mortalities, based on the upper 95% CI of density CRM predictions) then the population of the Alde-Ore Estuary SPA after 30 years will be 7-10.3% lower than it would have been in the absence of the additional mortality using the density independent model outputs and 2.1-3.2% lower using the density dependent model outputs. The population growth rate would be reduced by 0.3-0.4% using the density independent model and by 0.1% using the density dependent model (Table 11).

3.3.10. These values would be of some concern. However, Natural England does acknowledge that a breeding season apportionment rate of 30% is likely to be overly precautionary, given the proportion of the East Anglian LBBG population that the Alde-Ore Estuary SPA currently holds, and that there are other colonies (town colonies) located closer to Vanguard than the Alde-Ore. We note also note that even using the precautionary rate of 30% results in a collision prediction that only just exceeds 1% of baseline mortality (1.08%). On this basis, no AEOI for the LBBG feature of the Alde-Ore Estuary SPA can be concluded for collision impacts from Vanguard alone.

b. **In-combination collision risk impacts with other plans and projects**

3.3.11. As noted in our Deadline 7 response (REP7-075), we welcome that figures for the Hywind, Kincardine and Moray West offshore wind farms (OWFs) are included in the cumulative CRM table (and hence the in-combination assessment) for LBBG (Table 20 of AS-048).

3.3.12. As noted in our Deadline 7 response (REP7-075), we consider the approach taken by the Applicant for LBBG from the Alde-Ore Estuary SPA in paragraph 117 of REP6-021 for reaching an apportionment rate for in-combination in the non-breeding season of 4% is acceptable and note that this approach is again taken in AS-048. We also welcome that the Applicant has considered all offshore wind farms within 141km from the Alde-Ore in the breeding season assessment. However, the Applicant has again then applied a generic rate of 30% apportionment to the total breeding season collision predictions from all the wind farms within 141km of the Alde-Ore to apportion total in-combination collisions in the breeding season. As we have advised previously noted in REP2-038 and REP7-075, we consider this to be an overly simplistic approach, as this does not consider the distance of each of these wind farms from the Alde-Ore SPA, the other colonies within foraging range of each of these offshore wind farms, the size of each of the other offshore wind farms etc. This approach will potentially overestimate the contribution of some of the other projects and underestimate the contribution of others and the extent to which this underestimation of values is cancelled out by any overestimated values in the Applicant's calculated overall total is currently not known. We again suggest that the Applicant re-considers this issue. Potentially the most straightforward approach would be to use the apportionment rates used by the other wind farms in their assessments, as Natural England has advised for FFC SPA kittiwake, though other options might be appropriate and we would be happy to try to identify these with the Applicant.

3.3.13. We welcome that the in-combination assessment in AS-048 makes reference to the outputs from the updated LBBG Alde-Ore Estuary PVA outputs presented by the Applicant in REP7-063, but note the comments raised with regard to this PVA update set out in the section on impacts from Vanguard alone above.

3.3.14. We note an error in the Applicant's calculation in paragraph 150 of AS-048 for the number of in-combination LBBG collisions apportioned to the Alde-Ore Estuary in the breeding season. We calculate that 30% of the breeding season total of 63.3 birds for all wind farms within 141km of the Alde-Ore excluding Vanguard equals 19 birds. Then with the 2.6 birds apportioned to the Alde-Ore in the breeding season by the Applicant for Vanguard equals 21.6 (and not 19.9 as calculated by the Applicant). This gives an annual total of 21.6 in the breeding season plus 15 birds in the non-breeding season, which equals 37 birds per annum in-combination using the Applicant's apportionment rates for the Vanguard figures.

3.3.15. Natural England calculates the annual apportioned figure to the Alde-Ore Estuary SPA from Vanguard alone to be slightly higher than the Applicant at 5 collisions per year (4.7 in the breeding season and 0.3 in the non-breeding season) for using a precautionary 30% breeding season apportionment rate and the Applicants rates of 3.3% in the autumn and spring and 5% in winter for the worst case scenario layout of 2/3 of turbines in Vanguard West and 1/3 in Vanguard East. Using these figures gives a breeding season in-combination total of 19 + 4.7 = 23.7 and a non-breeding season in-combination total of 15. **This gives an annual in-combination total of 39 LBBG collisions per year**. Natural England notes that no collisions were apportioned to the Alde-Ore from Hornsea 3, which we are content with as the site is outside of the 141km foraging range of the Alde-Ore and no LBBG collisions were predicted in the non-breeding season. Both the Applicant's calculated in-combination figures and those calculated by Natural England equate to more than 1% of baseline mortality of the colony (see Table 12).

Table 12 Percentage of baseline mortality for in-combination collision impacts for LBBG for the Alde-Ore Estuary SPA. Baseline mortality calculated using adult only colony size and adult mortality rate (11.5% from Horswill & Robinson 2015). Note no collisions apportioned to Hornsea 3 in the in-combination assessment

LBBG PREDICTED IN-COMBINATION CRM MORTALITY, HRA: ALDE-ORE ESTUARY SPA		
	Mortality prediction	% of baseline mortality of Alde-Ore SPA* (2,000 pairs 2007-14, as used by Applicant)
Applicant's in-combination CRM, based on figures from Table 20 of AS-048	37	8.04
Natural England's calculated in-combination CRM, based on preferred apportionment rates and CRM figures for Vanguard	39	8.48

* 4,000 adults, 1% baseline mortality = 5 birds

3.3.16. Natural England has again focused our conclusions on the PVA outputs from the density independent model (see Table 13).

Table 13 Predicted population impacts on the LBBG population of the Alde-Ore Estuary SPA for the range of mortality impacts predicted for Norfolk Vanguard in-combination with other plans and projects. PVA impact metrics are as provided in REP7-063. The shaded cells represent the level of impact closest to the in-combination predictions in Table 12.

LBBG – ALDE-ORE ESTUARY SPA			
Additional mortality	% Baseline Mortality using population size of 4,000 adults (2007-2014), as used by the Applicant	Density Independent Model	
		Counterfactual of Final Population Size (CPS) after 30yrs – see Table 2 of REP7-063	Counterfactual of Growth rate (CGR) after 30yrs – see Table 3 of REP7-063*
35	7.61	0.775 (0.714-0.843)	0.991 (0.988-0.994)
40	8.70	0.748 (0.687-0.815)	0.990 (0.987-0.993)

* The Applicant has confirmed that the headings for the median and lower CIs are the wrong way around in REP7-063. So, we have presented the figures the correct way around above

3.3.17. If the additional mortality from the windfarm is 35-40 adults per annum (closest PVA outputs available in REP7-063 to Applicant's predicted 37 mortalities for the in-combination total and to the 39 in-combination total calculated by Natural England using our preferred approach for Vanguard alone figures) then the population of the Alde-Ore Estuary SPA after 30 years will be 22.5-25.2% lower than it would have been in the absence of the additional mortality. The population growth rate would be reduced by 0.9-1% (Table 13). If it is assumed that the population is stable then this would mean that the population would be 122.5-25.2% lower than the current population size. This would be counter to the restore conservation objective for this feature of the site. Vanguard's contribution to the in-combination total based on the Applicant's figures is 8.1%, whilst Vanguard's contribution to the in-combination total based on Natural England's figures is 12.8%.

3.3.18. It is not known what the growth rate of the colony will be over the next 30 years and this should be considered when judging the significance of predicted impacts against the conservation objectives for the feature.

3.3.19. As the Alde-Ore LBBG population is at best currently stable and the Applicant's PVA (REP7-063) suggests a baseline growth rate of -2% for the density independent model we have considered these levels of growth rates per annum. We have also considered a range of 1-5% growth rates per annum for if the colony may potentially grow in the future, although at present there seems considerable uncertainty regarding whether this can be achieved.

3.3.20. The Conservation Objective for the LBBG population of the Alde-Ore Estuary SPA is to restore the size of the breeding population to a level which is above 14,074 whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent.

3.3.21. If we assume a -2% per annum growth rate or a stable population then 35 or 40 additional mortalities per annum would result in the population declining below its current level and let alone be able to reach the target population of the conservation objective.

3.3.22. If we assume a 1% per annum growth rate then 35-40 additional mortalities per annum would result in the population being approximately 1,000-1,500 birds lower than without the additional mortality after 30 years and it would take over an additional 1,000 years to reach the target population compared to the no windfarm mortality scenario with 35 additional mortalities. But the population would never reach the target population of the conservation objective with an additional 40 mortalities.

3.3.23. If we assume a 2% per annum growth rate then 35-40 additional mortalities per annum would result in the population being approximately 2,000 birds lower than without the additional mortality after 30 years and it would take over an additional 50-60 years to reach the target population compared to the no windfarm mortality scenario.

3.3.24. If the LBBG population were to grow at a rate of 3% per annum over the next 30 years, then additional mortality of 35-40 per annum would result in the population being approximately 2,000-2,500 birds lower than without the additional mortality after 30 years and it would take over an additional 20 years to reach the target population compared to the no windfarm mortality scenario.

3.3.25. There is no evidence to suggest that the future population trend will be significantly different from the current trend, which is most likely to be stable, in which case there is a risk that the population could decline due to predicted mortality levels. Furthermore, given that the population is likely to be hindered from restoration to target levels even when more optimistic assumptions about the population trend of the colony are made, Natural England also considers that it is not possible to rule out AEOI even if the population starts to show modest growth.

3.3.26. **Therefore, as this feature has a restore conservation objective, and because there are indications that the population might even decline from current levels, Natural England advises that it is not possible to rule out AEOI of the LBBG feature of the Alde-Ore Estuary SPA for collision impacts from in-combination with other plans and projects**.

3.4. GREATER WASH SPA: LITTLE GULL

a. <u>Vanguard alone: collision impacts</u>

3.4.1. As noted in our Relevant Representations (RR-106), we agree with the Applicant's approach to apportioning of little gull collisions to the Greater Wash for Vanguard alone. Therefore, we agree with the Applicant's calculation in paragraph 187 of AS-048 of 0.6 collisions per annum apportioned to the Greater Wash SPA from Vanguard alone for the central collision prediction and that this equates to 0.24% of baseline mortality of the SPA population (using an SPA population of 1,255 and a mortality rate of 20% (calculated from the adult survival rate in Horswill & Robinson 2015 of 0.8).

3.4.2. We note that the Applicant has not considered the uncertainty/variability in the CRM input parameters in the assessment through considering the range of predicted impacts resulting from use of the 95% Cis of the density estimates. The range of figures for EIA alone for little calculated by Natural England is 0-12 birds. Therefore, for the upper 95% CI of density and using the Applicant's approach to apportionment, a total of 1.5 collisions is apportioned to the Greater Wash SPA. This equates to 0.6% of baseline mortality of the SPA population.

3.4.3. **Based on this, we agree with the Applicant's conclusion in paragraph 188 of AS-048 of no AEOI of the little gull feature of the Greater Wash SPA for collision impacts from Vanguard alone.**

b. **<u>In-combination collision risk impacts with other plans and projects</u>**

3.4.4. We welcome that the assessment in AS-048 now includes an in-combination assessment for collision risk to little gull from the Greater Wash SPA. We note that the Applicant has included Triton Knoll, Race Bank, Sheringham Shoal, Hornsea 1, Hornsea 2, Hornsea 3 and Vanguard in Table 25 of AS-048 as OWFs considered to have connectivity with the Greater Wash SPA.

3.4.5. Clarification is required on the approach the Applicant has taken to decide whether sites have connectivity with the Greater Wash SPA. Natural England would also advise that:

- Dudgeon is also included in the in-combination assessment (note that Dudgeon has completed an assessment of collision risk for little gull at the Greater Wash SPA).
- As Vanguard is included in the in-combination assessment, we also query why the other projects in the former East Anglia zone (e.g. East Anglia 1 and East Anglia 3) are also not included.

3.4.6. We agree that the CRM figures presented for the various sites in Table 25 of AS-048 have been updated for an avoidance rate of 99.2%.

3.4.7. As noted in our general comments above on cumulative/in-combination assessments, we do not consider it is appropriate to adjust the figures for the other OWFs based on build out capacities unless the reduction is legally secured and CRM re-run.

3.4.8. As we do not consider that figures have been included in the assessment from all relevant OWFs we cannot reach a conclusion regarding the impacts of in-combination collisions on the little gull feature of the Greater Wash SPA.

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