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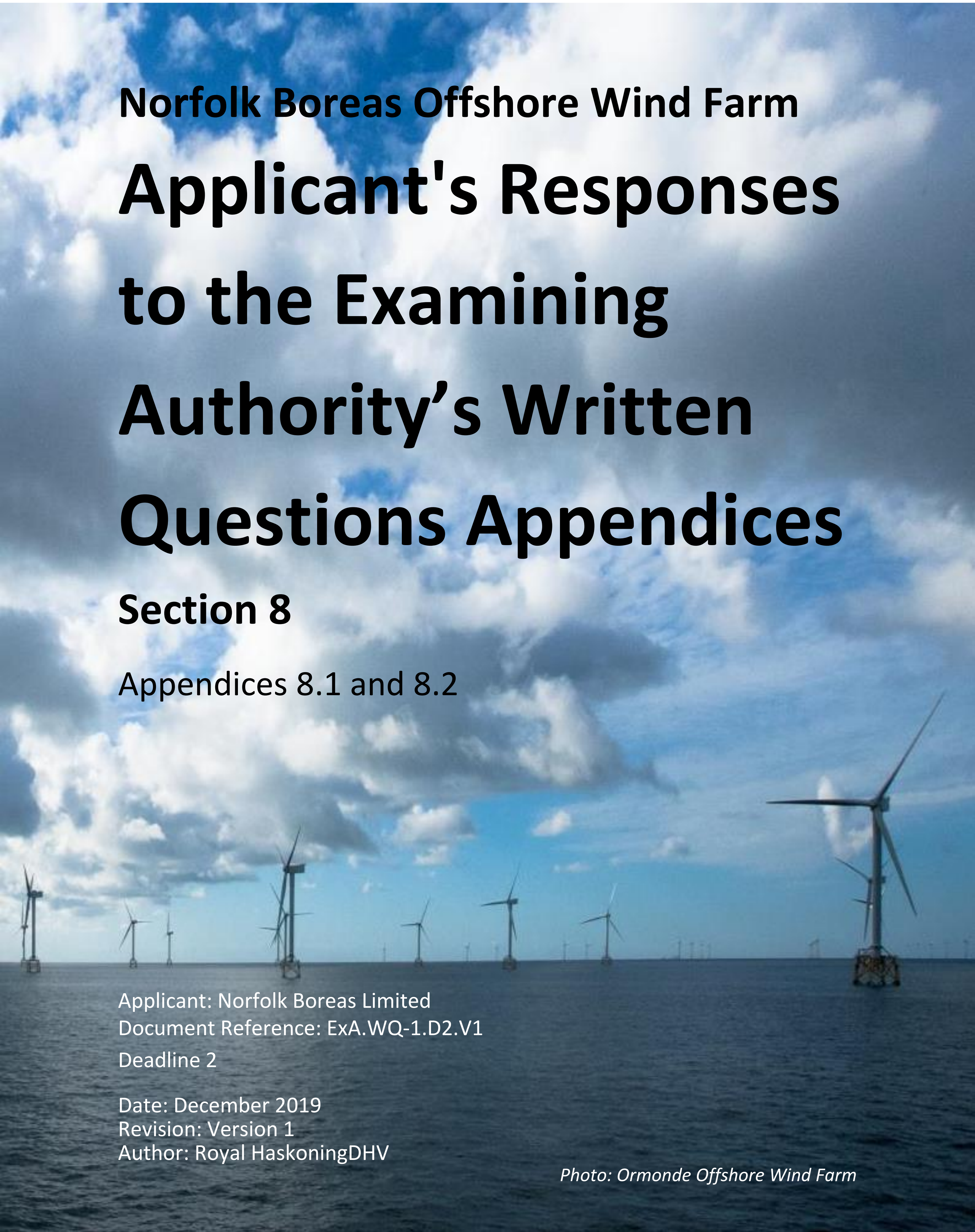

Norfolk Boreas Offshore Wind Farm

Applicant's Responses to the Examining Authority’s Written Questions Appendices

Section 8

Appendices 8.1 and 8.2

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

Date: December 2019
Revision: Version 1
Author: Royal HaskoningDHV

Photo: Ormonde Offshore Wind Farm

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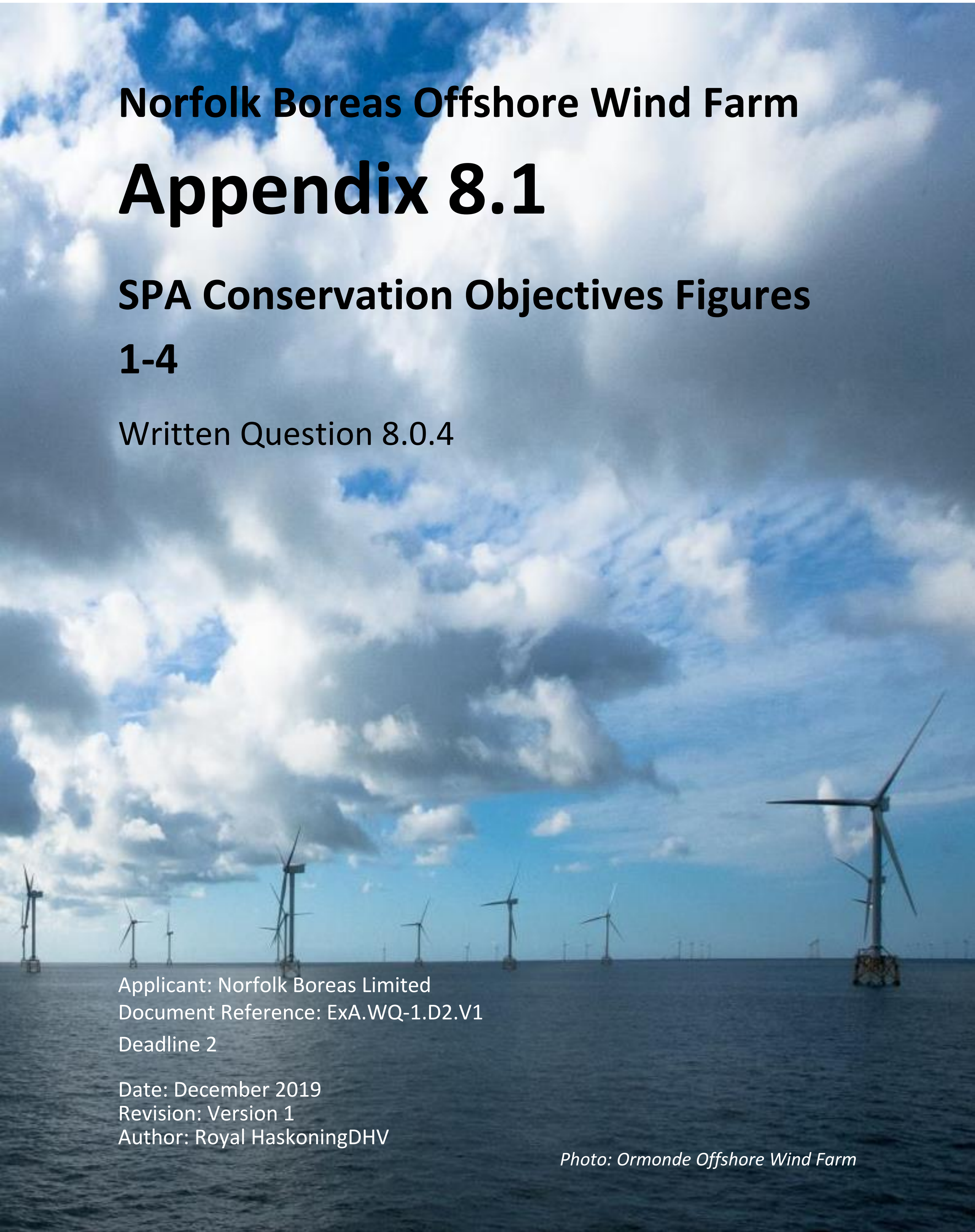


Norfolk Boreas Offshore Wind Farm

Appendix 8.1

SPA Conservation Objectives Figures 1-4

Written Question 8.0.4

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

Date: December 2019
Revision: Version 1
Author: Royal HaskoningDHV

Photo: Ormonde Offshore Wind Farm

European Site Conservation Objectives for Breydon Water Special Protection Area

Site Code: UK9009181

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the ‘Qualifying Features’ listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A037	*Cygnus columbianus bewickii*; Bewick’s swan (Non-breeding)
A132	*Recurvirostra avosetta*; Pied avocet (Non-breeding)
A140	*Pluvialis apricaria*; European golden plover (Non-breeding)
A142	*Vanellus vanellus*; Northern lapwing (Non-breeding)
A151	*Philomachus pugnax*; Ruff (Non-breeding)
A193	*Sterna hirundo*; Common tern (Breeding)A

Waterbird assemblage

This is a European Marine Site

This SPA is a part of the Breydon Water European Marine Site (EMS). These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

European Site Conservation Objectives for Broadland Special Protection Area
Site Code: UK9009253

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the ‘Qualifying Features’ listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A021	*Botaurus stellaris*; Great bittern (Breeding)
A037	*Cygnus columbianus bewickii*; Bewick’s swan (Non-breeding)
A038	*Cygnus cygnus*; Whooper swan (Non-breeding)
A050	*Anas penelope*; Eurasian wigeon (Non-breeding)
A051	*Anas strepera*; Gadwall (Non-breeding)
A056	*Anas clypeata*; Northern shoveler (Non-breeding)
A081	*Circus aeruginosus*; Eurasian marsh harrier (Breeding)
A082	*Circus cyaneus*; Hen harrier (Non-breeding)
A151	*Philomachus pugnax*; Ruff (Non-breeding)

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) (‘the Habitats Regulations’). They must be considered when a competent authority is required to make a ‘Habitats Regulations Assessment’ including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

European Site Conservation Objectives for North Norfolk Coast Special Protection Area
Site Code: UK9009031

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A021 *Botaurus stellaris*; Great bittern (Breeding)

A040 *Anser brachyrhynchus*; Pink-footed goose (Non-breeding)

A046a *Branta bernicla bernicla*; Dark-bellied brent goose (Non-breeding)

A050 *Anas penelope*; Eurasian wigeon (Non-breeding)

A081 *Circus aeruginosus*; Eurasian marsh harrier (Breeding)

A084 *Circus pygargus*; Montagu's harrier (Breeding)

A132 *Recurvirostra avosetta*; Pied avocet (Breeding)

A143 *Calidris canutus*; Red knot (Non-breeding)

A191 *Sterna sandvicensis*; Sandwich tern (Breeding)

A193 *Sterna hirundo*; Common tern (Breeding)

A195 *Sterna albifrons*; Little tern (Breeding)

This is a European Marine Site

This SPA is a part of The Wash and North Norfolk Coast European Marine Site (EMS). These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 30 June 2014 to reflect the consolidation of the Habitats Regulations in 2017.

European Site Conservation Objectives for Outer Thames Special Protection Area
Site Code: UK9020309

With regard to the SPA and the individual species and/or assemblage of species for which the site has been classified (the 'Qualifying Features' listed below), and subject to natural change;

Ensure that the integrity of the site is maintained or restored as appropriate, and ensure that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring;

- **The extent and distribution of the habitats of the qualifying features**
- **The structure and function of the habitats of the qualifying features**
- **The supporting processes on which the habitats of the qualifying features rely**
- **The population of each of the qualifying features, and,**
- **The distribution of the qualifying features within the site.**

This document should be read in conjunction with the accompanying *Supplementary Advice* document, which provides more detailed advice and information to enable the application and achievement of the Objectives set out above.

Qualifying Features:

A001 Gavia stellata; Red-throated diver (Non-breeding)
A193 Sterna hirundo; Common tern (Breeding)
A195 Sternula albifrons; Little tern (Breeding)

This is a European Marine Site

This SPA is a part of the Outer Thames European Marine Site (EMS). These Conservation Objectives should be used in conjunction with the Conservation Advice document for the EMS. Natural England's formal Conservation Advice for European Marine Sites can be found via GOV.UK.

Explanatory Notes: European Site Conservation Objectives

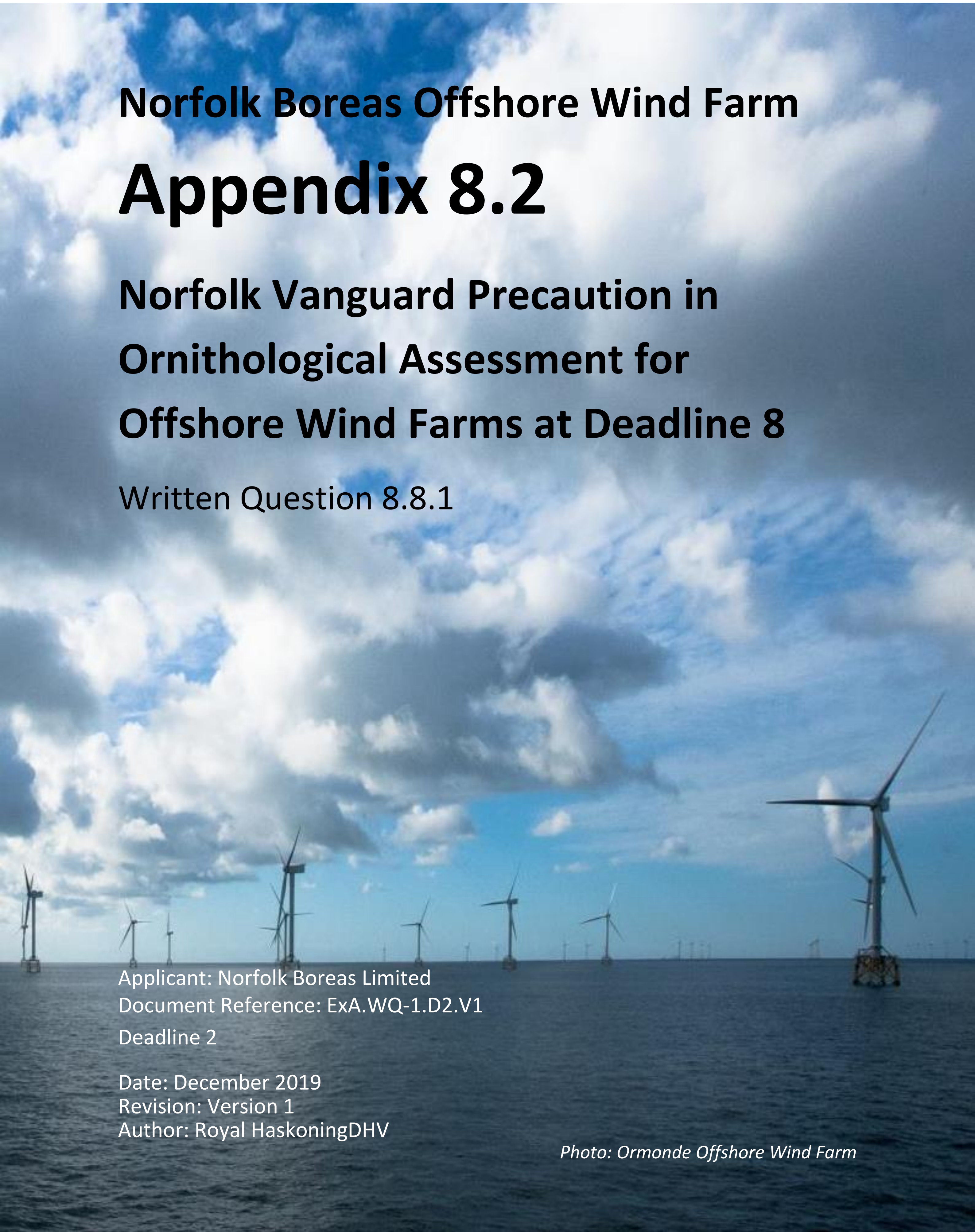
These Conservation Objectives are those referred to in the Conservation of Habitats and Species Regulations 2017 (as amended) ('the Habitats Regulations'). They must be considered when a competent authority is required to make a 'Habitats Regulations Assessment' including an Appropriate Assessment, under the relevant parts of this legislation.

These Conservation Objectives, and the accompanying Supplementary Advice (where this is available), will also provide a framework to inform the management of the European Site and the prevention of deterioration of habitats and significant disturbance of its qualifying features

These Conservation Objectives are set for each bird feature for a Special Protection Area (SPA).

Where these objectives are being met, the site will be considered to exhibit a high degree of integrity and to be contributing to achieving the aims of the Wild Birds Directive.

Publication date: 21 February 2019 (version 3). This document updates and replaces an earlier version dated 20 December 2017 to reflect the consolidation of the Habitats Regulations in 2017.



Norfolk Boreas Offshore Wind Farm

Appendix 8.2

Norfolk Vanguard Precaution in Ornithological Assessment for Offshore Wind Farms at Deadline 8

Written Question 8.8.1

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

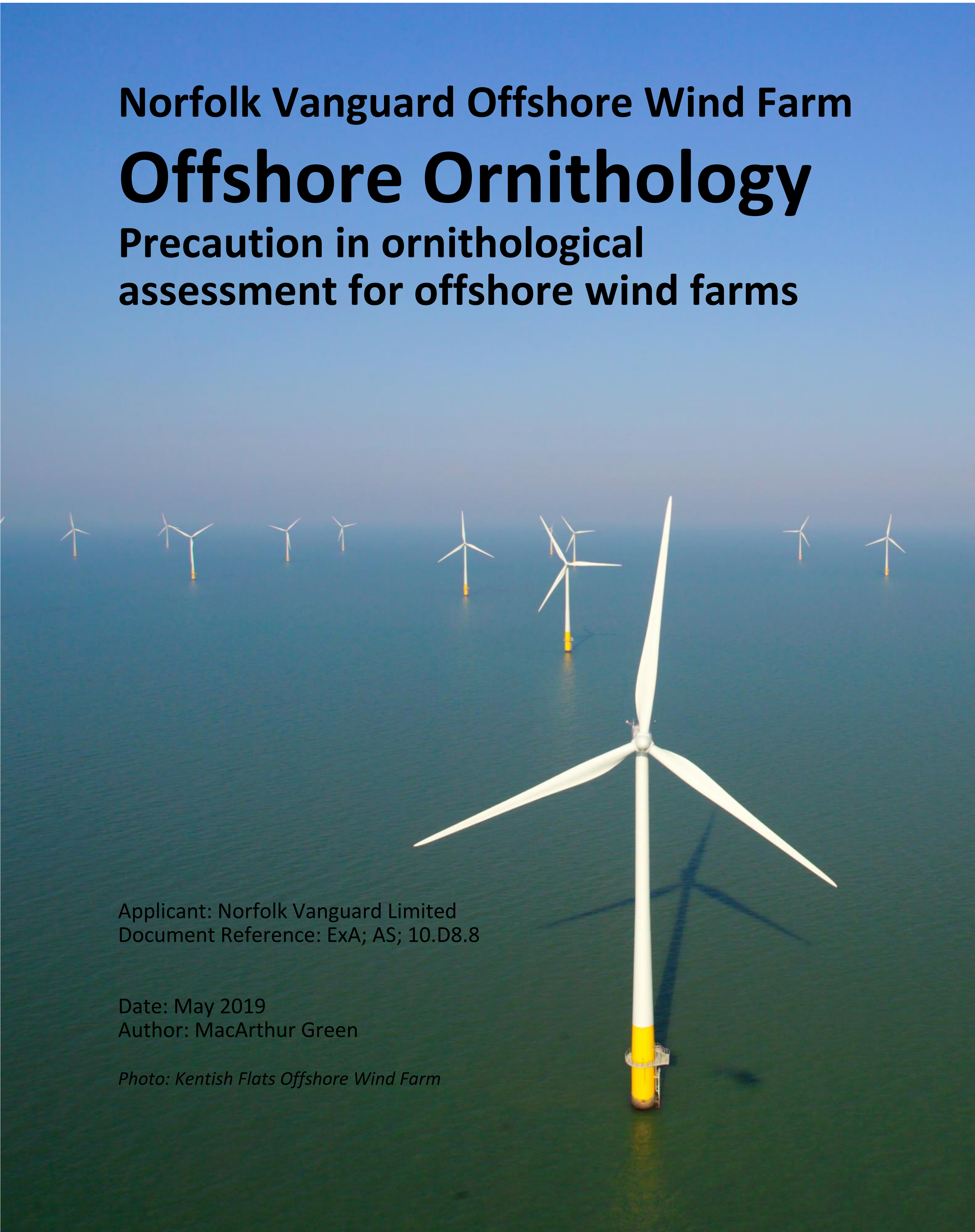
Date: December 2019
Revision: Version 1
Author: Royal HaskoningDHV

Photo: Ormonde Offshore Wind Farm

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

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[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.

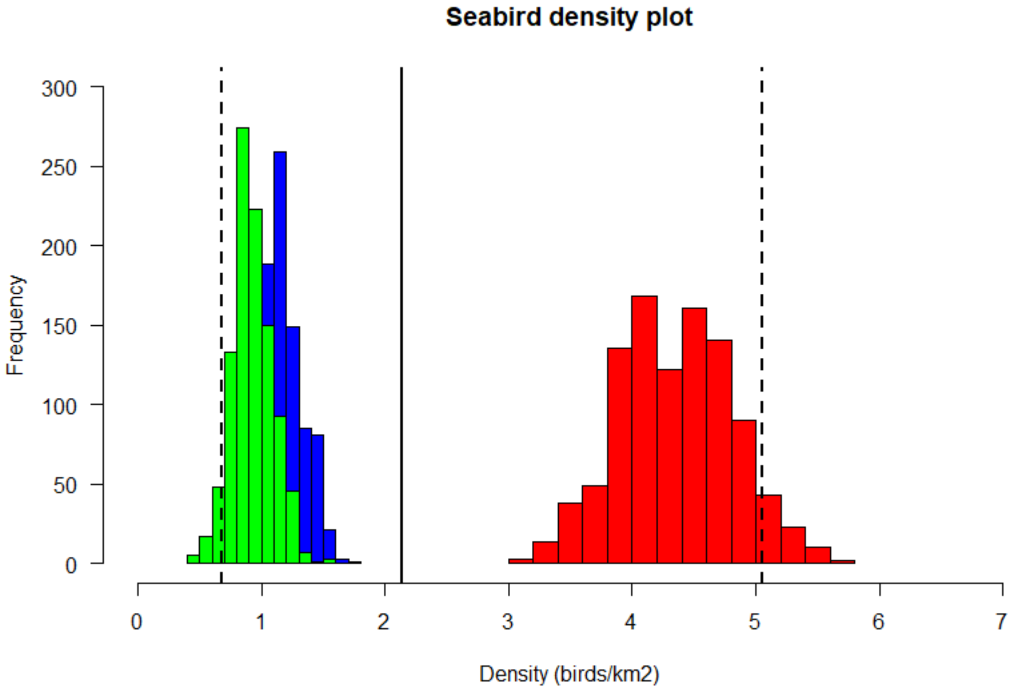

Figure 1. Gannet density data. Each coloured histogram is the bootstrapped sample for November (red: 2012, blue: 2013 and green: 2015) for gannets recorded in flight in surveys of Norfolk Vanguard East. The vertical black lines are the overall mean density (solid) and the 95% confidence intervals (dashed) 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.

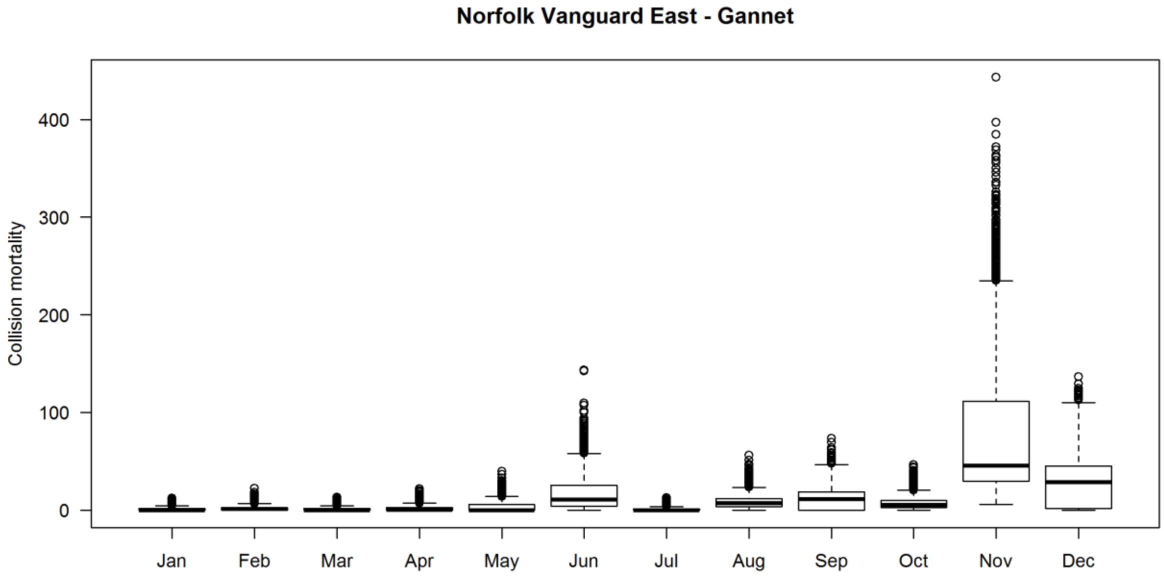

Figure 2. Norfolk Vanguard East, Gannet Option 2 collision mortality estimates calculated with stochasticity in seabird density, avoidance rate, flight height and nocturnal activity. Solid bars are the median, boxes indicate the 50% range, whiskers the 95% range and circles are outliers. (Note

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[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

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[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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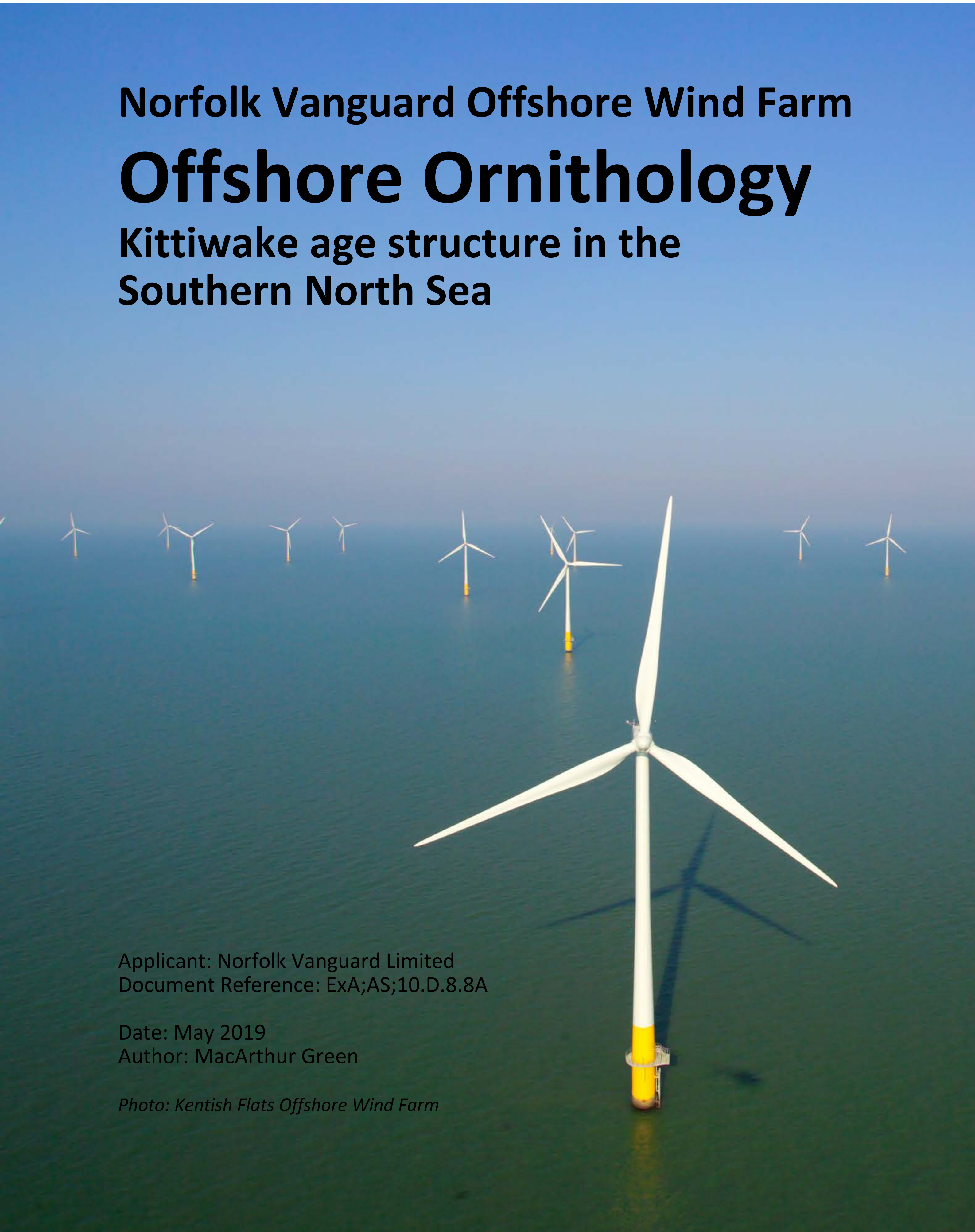

Norfolk Vanguard Offshore Wind Farm

Offshore Ornithology

Kittiwake age structure in the Southern North Sea

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;AS;10.D.8.8A

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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).

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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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